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16. ABSTRACT

The United States-Japan Ministerial Agreement of May 1974 provided for the exchange of environmental information in several areas of mutual concern. This report is the compilation of papers presented at the Fourth U.S.-Japan Experts' Meeting on the Management of Bottom Sediments Containing Toxic Substances, one of the 10 identified problem areas.

The first meeting was held in Corvallis, Oregon in November 1975. The second meeting was hosted by the Japanese Government in October 1976. The third session was convened in November 1977 in Easton, Maryland and the fourth session (at which these papers were presented) in Tokyo.

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Management of Bottom Sediments Containing Toxic Substances

Proceedings of the Fourth U.S.—Japan Experts' Meeting
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MANAGEMENT OF BOTTOM SEDIMENTS CONTAINING TOXIC SUBSTANCES

Proceedings of the Fourth U.S.-Japan Experts' Meeting
October 1978--Tokyo, Japan

edited by

Spencer A. Peterson and Karen K. Randolph
Corvallis Environmental Research Laboratory
Corvallis, Oregon 97330

CORVALLIS ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CORVALLIS, OREGON 97330

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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL).

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lakes and streams; and the development of predictive models on the movements of pollutants in the biosphere. In May 1974 the United States-Japan Ministerial Agreement provided for the exchange of environmental information on several areas of mutual concern. This report is the compilation of papers presented at the Fourth U.S.-Japan Experts' Meeting on the Management of Bottom Sediments Containing Toxic Substances which was held October 30-November 3, 1978 in Tokyo.

Thomas A. Murphy
Director, CERL

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DREDGING AND TREATMENT OF SEDIMENTS IN THE PORT OF TAGONOURA

Toji Inada
Chief, Port and Harbor Section, Civil Engineering Division
Shizuoka Prefectural Government, Japan

INTRODUCTION

The paper industry in the Gakunan district near the Port of Tagonoura has prospered by using the extensive water and forest resources at the foot of Mt. Fuji. The industry began manufacturing paper about 90 years ago and has expanded through mechanization and integration of smaller businesses.

After World War II, many enterprises such as Nissan Motor Company, Toshiba Electric Company, and Asahi Chemical Industry Company, built large factories in this district. Now there are 1,300 factories producing paper, pulp, foods, machines and metal merchandise. The opening of the Port of Tagonoura in 1961 improved cargo transportation. Construction of the man made port started in 1958. It accomodates 10,000 ton-class ships.

The industrial output in this district is 950 billion yen, of which 47% is produced by the paper, pulp and paper-processing industry — the largest such group in Japan. The factories use about 2.0×10^6 m^3 of water a day. Sewage drains into the Port of Tagonoura via the Numa River, the Urui River and the Gakunan drainage basin. Suspended solids (SS) from the sewage accumulate on the bottom of the port.

The sewage sludge is called "hedoro" in Japanese. In 1970, 1.2×10^6 metric tons of Hedoro was discharged into the environment. The sludge spread and interfered with port functions. It produced toxic H_2S gas. The odor became a public nuisance.

The first management effort, in 1971, was to remove the sludge. The plan was to use self-propelling barges to dump the sludge in the open sea, 320 km from the Port of Tagonoura. The barge hulls were reinforced and remodeled to carry sludge.

Before the plan could be activated it was opposed by fishermen and scientists who had attended the World Oceanography Congress, and therefore the plan had to be changed. On-land treatment was the only alternative.

On-land treatment creates problems on how to dehydrate the sediments and avoid secondary public nuisances from unpleasant odors, toxic gases produced by the dehydration process and contamination of groundwater by infiltration of polluted water. The dry bed of the Fuji river was selected as a treatment site since it had extensive area available for mass dehydration.

The bottom sediments were first dredged using a cutterless suction dredge to minimize sediment agitation. Sediments were loaded on a barge and carried to the mouth of the Fuji River where they were pumped to a dehydration pond via pipeline.

With this system the first management effort dehydrated 110,000 tons of sediment at a cost of 820 million yen. Since the capacity of the dehydration pond had been determined, the second and third mass management effort used a bigger suction dredge.

The second and third management efforts were implemented from 1972 to 1974. The sediment was dredged by a 5,000 ps suction dredge and carried directly to the dehydration pond on the dry bed of the Fuji River through a pipe 6 km long. Natural dehydration then occurred.

The system was continually improved. Steel-covers were installed on the upper part of the outside of the cutter so the sediment would not be dispersed through agitation. Fishing nets were used to cover the dredging area to avoid contamination of the sea. The pipes for carrying sediments were pressure tested at 15 kg/cm^2 which exceeded the maximum pressure of the dredge pump. Pressure gauges were installed at each kilometer of pipe and readings were monitored and telephoned to the dredge tender.

A $Ca(OH)_2$ solution of 1,500 ppm was injected into the pipe during transit to remove H_2S. A $FeCl_3$ solution (39%) of 333 ppm was added to adjust the pH value. A high molecular cohesion agent of 10 ppm was added to accelerate dehydration. Finally, an aeration tank was installed at the pipe outlet to remove any remaining H_2S gas.

The sediment's water content was about 98% before dehydration and was reduced to about 80% after natural drying in the dehydration pond. The dehydrated sediments were reclaimed after mixing with gravel and cobblestones from the dry river bed in a ratio of one to one. This was done with a bulldozer.

A total of 1.3×10^6 m^3 of sediment was removed by these three management efforts. During the same period, effluent standards were established to control the source of wastewater. These standards were raised four times. As a result, new accumulations of sludge have decreased, however 5.2×10^5 m^3 of sediment still remained in 1975. If the sediment had been left as it was, pollution would recur. Therefore, a fourth sediment management effort was scheduled to be implemented by the Public Nuisance Countermeasure Council.

This paper reports on the fourth management effort in the Port of Tagonoura.

FOURTH MANGEMENT EFFORT

PROPERTIES OF THE SEDIMENT

The properties of the sediment were: muddy sludge, 20.4% average ignition-loss (ash-free dry weight), 70-80% water content, an infinitesimal quantity of heavy-metal, and an average value of 28.5 ppm of PCBs. The PCB content had been decreased by the more stringent regulation of water quality and the prohibition on using PCB-containing copy paper as toilet tissue. The PCB could not be detected by an elution test. Taking all matters into consideration, the plan was to remove only the sediment containing PCB in excess of 10 ppm or with an ignition loss of more than 15%.

PLANNING

Since on-land management was mandatory, the adjacent right fork of the dry Fuji River and a timber pond in Port Tagonoura were selected as the landfill areas. The timber pond was chosen because the sediment could be used as fill material in a separate project to remodel the pond into a timber depot. The right folk of the dry river bed was selected for two reasons. First, reclamation of the sediment in the left fork of the dry river bed had proven to be safe and effective. Second, a sporting arena to be constructed on the reclaimed ground of the left fork had gained consideable public favor.

Engineering methods were studied which could do this project inexpensively, effectively and safely, and without creating any secondary public nuisance factors.

A big suction dredge and natural dehydration was considered. But, the system was discarded for the following reasons. First, the right fork of the river bed was narrower than the left fork and, consequently, limited the space for dehydration ponds, thus restricting the working hours of the big suction dredge and lowering the efficiency of the whole system. Second, the dredge discharge pipe was vulnerable to flood damage since it would have to be installed across the river. Third, the system carried a risk of polluting water supplies near the right fork of the dry riverbed. Fourth, the natural dehydration method could not be used in the timber pond area since the pond contained too much water and noxious odors from the sediment would pollute the residential zone around the pond.

Next, methods of mechanical dehydration instead of natural dehydration were examined. These methods had been widely used in the wastewater disposal plants of the paper factories. But, this method was found unsuitable by experiments which showed that the equipment would be corroded by seawater and screens broken by materials such as vinyl, wire, blocks of wood, or any earth and sand carried downstream from large washouts near Mt. Fuji.

METHOD OF TREATMENT

Because of the above problems, a system using rented machinery requiring little capital investment was adopted. This method picked up highly-polluted sediments with a grab dredge and added agents to dehydrate and solidify them. We started to look for processing agents for this system.

Quicklime was first considered because it has been used successfully to strengthen weak soils. Quicklime absorbs 32% of its weight in water and quicklime-enforced sludge is stable over time. Results of these experiments showed that a 72.5% water content in the sediment decreased linearly to 62.2%

10 minutes after the the addition of 5% quicklime and to 5.9% 10 minutes after the addition of 10% quicklime, and that the addition of 20% quicklime solidified the sediment enough to make it suitable for reclamation soil.

Although the problem of controlling lime dust and vapor produced by the reaction was trivial in the laboratory, the problem could not be ignored for large scale use involving several tons of quicklime. Different approaches were considered, including mixing with scrapers in enclosed barge and mixing under an air curtain created by a power blower installed on the barge. No completely satisfactory perfect answer was found. In addition to this, there was another problem. Mixing of the quicklime solution had to take place very rapidly because of the tendency for quicklime to solidify immediately.

As a result of the number and complexity of these problems, quicklime was discarded as a hardening agent. When selecting the proper agent, the following characteristics were considered:

1. The solidifying agent must strengthen the mixed sediment to more than 1.0 kg/cm^2 of uniaxial compression strength (referred to as UCS) so it can be worked by a swamp bulldozer during reclamation. Although the ordinary load for a swamp bulldozer is 0.3 kg/cm^2, a standard of 1.0 kg/cm^2 was adopted for safety.

2. It must be easily mixed.

3. It should not produce any dust or fumes during mixing.

4. No harmful substance should be detectable and no secondary public nuisance factors should be created by it.

5. It must be economical.

More than 1500 combinations of 40 kinds of material were tested, including solicic acid soda, volcanic ash, cements, plasters, incineration ash, and some dehydration agents developed by private companies. A combination of 12% cement, 5% incineration ash and 4% exhaust-gas-desulfurization-plaster (EGDP) was selected. The following three ingredients - aluminum in the ash (Al_2O_3, 50~58%), calcium in the EGDP ($CaSO_4 \cdot 2H_2O$) and calcium in the cement - dehydrate and promote solidification of the high-water content sediments.

Incineration ash can be produced by nearby paper factories. The EGDP could be obtained from factories at the port. Both of these are waste and can be purchased inexpensively. Later research determined that good quality incineration ash to meet environmental standards could not be produced in sufficient volume. As a result, fly ash cement [referred to as FA (B-type)] was used instead of incineration ash and it was necessary to increase its volume to 13% to make the required UCS (Figure 1).

The UCS of the sediment 48 hours after the addition of 4% EGDP and 13% of various kinds of cements showed good values of 0.8 kg/cm^2 for FA (A-type) and 1.12 kg/cm^2 for FA (B-type), which are better than the value of 0.68 kg/cm^2 for normal Portland cement. Consequently, it was found that including a

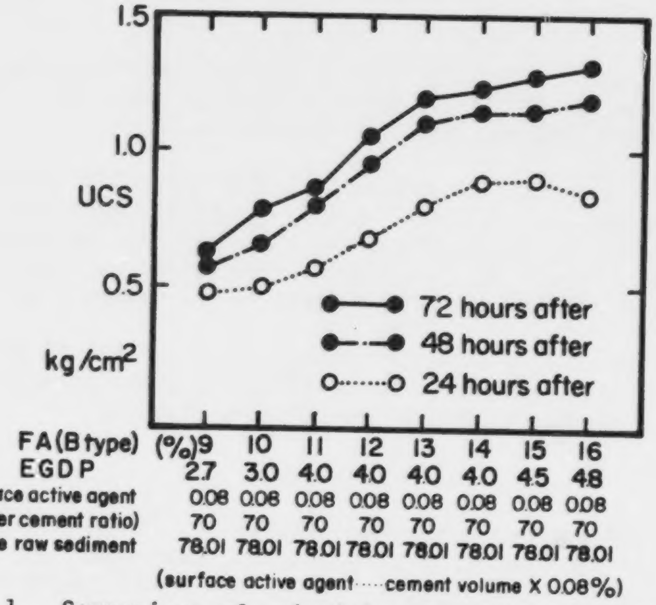


Figure 1. Comparison of uniaxial compression strength (UCS) for various combinations.

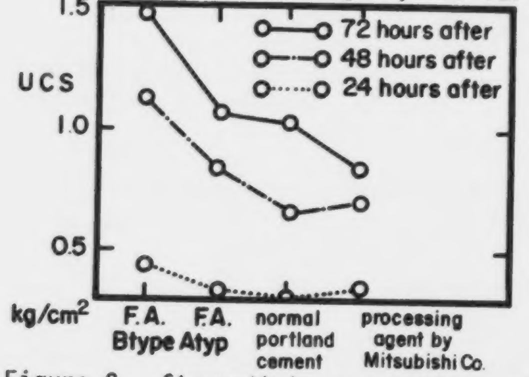


Figure 2. Strength by annexations of various kinds of cements.

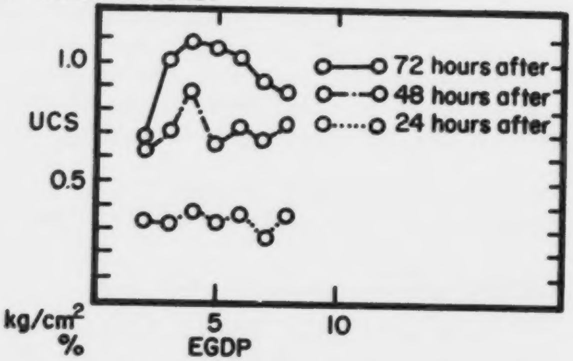


Figure 3. Strength by the annexation of EGDP.

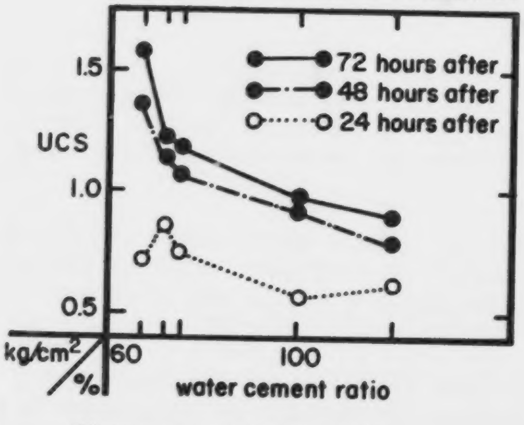


Figure 4. UCS for W/C

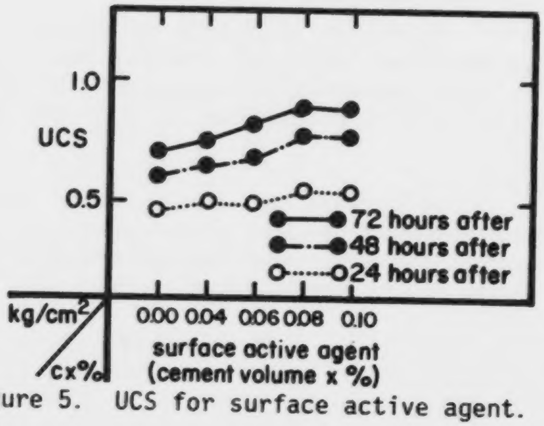


Figure 5. UCS for surface active agent.

suitable quantity of the fly ash solidifys and strengthens sediment which contains a large quantity of paper sludge (Figure 2). The most effective mixing ratio of the EGDP was found at 4% (Figure 3).

To add the agents, the cement and EGDP at a 70% water-cement ratio was mixed with the surface active agent to promote hydration. The water-cement ratio was selected by considering the efficiency of machinery, reactivity and dust prevention (Figure 4). Compared to no additive, the UCS was doubled by the addition of 0.08% surface active agent to the cement's volume (Figure 5).

These experiments showed the combinations of 13% FA (B-type) and 4% EGDP to the weight of the sediment, and 0.08% surface active agent to the weight of the cement was the proper combination for this program.

IMPLEMENTATION

Ideally, the methodology works like this: The sediment dredged by a grab dredge should be carried by a barge to the quay wall, and it should then be mixed with the processing agents. Next, the mixed sediment should be hauled by dump trucks to the reclamation site (Figure 6).

The processing volume per day was planned at about 1,100 m^3 based on considerations of economics and efficiency; namely, the holding capacities of the temporary depots, the mixing plant and the spillwater-processing apparatus as well as the impacts on neighboring activities created by the noise, traffic tieups caused by dump trucks, etc.

The management project in 1977, where sediment was reclaimed only in the timber pond, was conducted as described below:

Dredging Work

A 6 m^3 bucket dredge enabled the management of 1,100 m^3 sediment per 8-hour day in a program cycle of dredging, mixing and temporary detention. Four closed-bottom box type barges with 500 m^3 of loading capacity were used. This allowed for a 20% volume increase after mixing the processing agents into 350 m^3 of dredged sediment.

To prevent pollution of nearby areas caused by turbulence and release of sludge from the grab, control measures were selected carefully. This problem is most important in management of secondary public nuisance factors. Therefore, a silt curtain was laid in the sea surrounding the dredging site (Figure 6).

The curtain was made of soft vinyl-covered canvas which could withstand the range of tides. Steel had been considered as a curtain material but it was discarded since the height could not be changed at the ebb tide and it could not be easily moved.

The 50 kg anchors with chain ballasts were installed at the lower end of the curtain to enable the curtain to conform to the contour of the uneven sea floor. Since the curtain is lowered as the sea-bottom is deepened by dredging, seawater inside the curtain cannot flow to the outside.

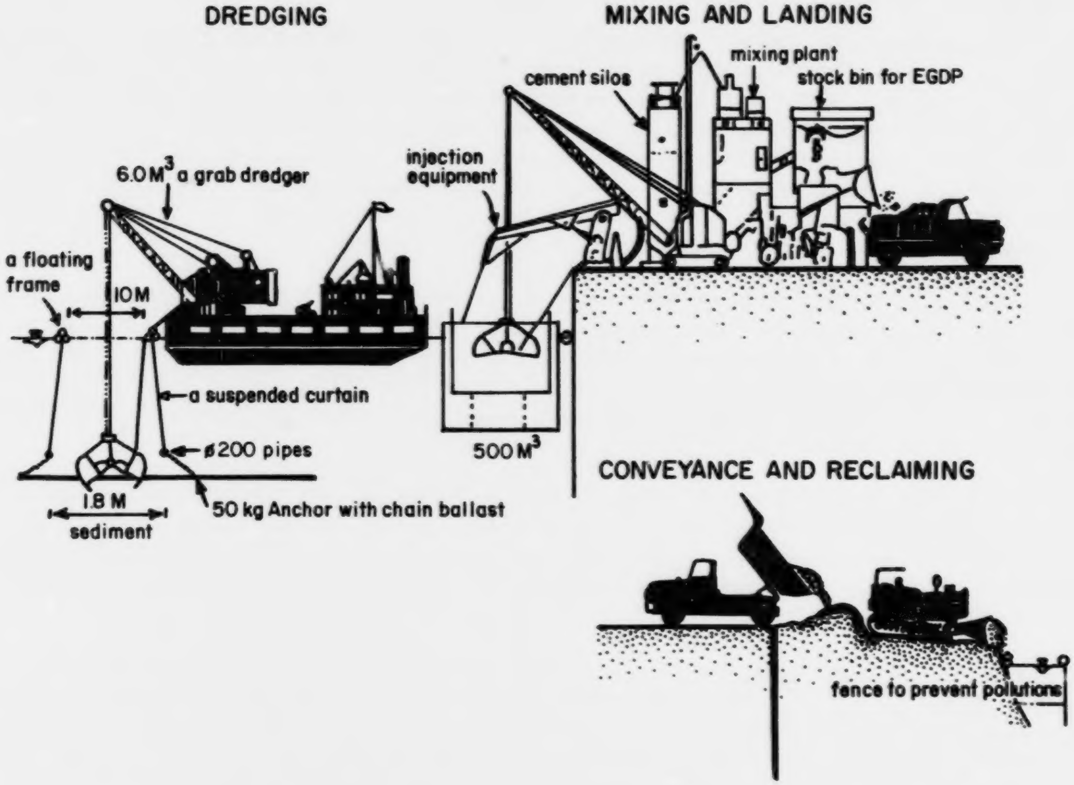

DREDGING
MIXING AND LANDING
mixing plant
cement silos
stock bin for EGDP
injection equipment
6.0 M^3 a grab dredger
a floating frame
10M
a suspended curtain
ø200 pipes
500 M^3
1.8 M
sediment
50 kg Anchor with chain ballast
CONVEYANCE AND RECLAIMING
fence to prevent pollutions

Figure 6. Treatment program.

The shape of the vertical section of the curtain was a trapezoid: 10 m for the upper part, 18 m for the middle and 28 m for the lower part. Approximately 500 m^3 of sediment was expected to be dredged from each curtain shrouded site. Two sets of curtains and frames were constantly employed. The curtain was moved to the next site after dredging was completed at the initial site. Each curtain was moved only after adding 3 ppm of coagulation agent to precipitate the suspended solids inside the curtain.

No abnormal signs were observed at observation points around the curtain during dredging. The curtain effectively prevented pollution. Emission of the offensive odor of H_2S gas was negligible because the dredging was done during the cool season. Also, hypochlorous acid soda was sprayed for desulfurization and deodorization. As a result, no air was polluted.

The average dredging time required to load 350 m^3 of sediment onto a barge was 2½ hours. This entailed 90 grabs. The efficiency of the grab was good because the water content of sediment decreased as dredging depth increased.

Mixing the Sediments with the Processing Agents

The processing agents were mixed in a plant installed behind the quay wall. The mixing process was mechanized and controlled at a single station. In the process, EGDP and water were combined to allow a 70% water-cement ratio. Then cement and activator were weighed and mixed in by a 1-m^3 turbomixer. Next, the mixed agents were injected into the sediment while still in the barge. The time required from weighing to injection was about one minute. The mixing could be done almost continuously because every batch was removed by pump after temporary detention in the storage chambers.

Since the water content varied with every batch of dredged sediment, and since the water-cement ratio affects the UCS, it is necessary to quickly determine how much cement to add to the mix. The water-cement ratio was determined from the water-content value measured either by a Kett moisture-measuring instrument or by comparison with the water content of sediments gathered in test dredging.

As a result of testing, it was proven feasible to be within 65% of the average water-cement ratio since the precise value of the latter measurement was close to the measured value and also since the water content had been homogenized by the narrowness of the dredging area and by drainage of excessive water from the barge.

Two clamshells were used to mix the sediment with the solidifying agents on the barge. The mixing took about 2 hours per barge (350 m^3). This time was determined from data on the strength of the mix relative to the frequency of mixing, as found by experiments and observations at the job site. The value of the UCS measured at the job site after 48 hours was 0.3~1.0 kg/cm^2. Soft sediments were strengthened by extending the temporary detention time.

The mixed sediment was temporarily detained for 24 hours in the barge, and again detained for 24 hours after unloading. The sediment was then carried to the reclamation area. The vicinity was free of pollution because the emission of noxious odors and H_2S gas was minimized by the mixing.

After some accidents occurred where the bottom of the barge was damaged by an unskilled grab operator, it was necessary to reinforce the bottoms of the barges.

Reclamation Work

Our policy was to avoid exudation of sediments. The open part of the timber pond was coffered with steel sheet piles, back-filled with fine sand and covered with vinyl sheets. All the levees of the pond were covered with vinyl sheets even though they were considered safe.

It was estimated that pollution of groundwater would not occur because: The bottom of the pond had a good clay layer; no percolation happened in the experiments; the sediment to be reclaimed was safe because no harmful substances had been detected in it; and the sediment would be gradually solidified with chemical agents. To make doubly sure, several wells were sunk around the pond and the quality of the underground water was monitored throughout the reclamation project.

The sediments were pushed out from a side levee using a swamp bulldozer, because the sediments should settle in the water of the pond. The water surface was covered with vinyl sheets to prevent the escape of offensive odors and H_2S gas. Some hypochlorous acid soda was sprayed under the sheets to neutralize odors and gas.

The draining of the wastewater from the pond after treatment was done concurrently with progress of the reclamation project. A settling pond 5 m by 125 m was built in the pond. About 2.0 ppm of cohesion agent and 500 ppm of $FeCl_3$ (to ajust the pH value) were injected into the wastewater at the inlet. The solid materials in the wastewater coagulated and settled in the main pond and the water was then drained outside the pond after rapid filtration through a sand bed.

Water quality of the outflow was constantly monitored by measuring transparency. Acceptable values were determined by the correlation of experimental values of transparency relative to SS and PCB. No elution of PCB was detected by later analysis. This demonstrated the efficacy of the methods.

Some sediment from the bottom of the timber pond was brought to the surface during the reclamation. We treated the sediments by mixing 13% cement, 4% EGDP and 0.08% activator.

The reclaimed ground was soon available as a timber depot because its firmness had increased from 3.9~13/0 kg/cm^2 after only 2 months. Work was done from March, 1977 to May, 1977 and from October, 1977 to December, 1977, thus avoiding the production of gases in summer. In this project 15,000 m^3 of sediment was reclaimed in the timber pond at a cost of 790 million yen, of which 2,160 yen per 1 m^3 of sediment was for the cement and chemical agents. The cost does not include the expenses of bank protection and finishing the reclaimed ground.

Supervision Plan

This work was supervised under the provision of "A Tentative Guide to the Management of Sediments" issued by the Director General of the Environmental Agency of Japan. Basic observation stations were established at the mouth of the port to monitor seawater quality. A supplementary observation station was set up to allow estimates of water quality changes at the main observation station. This permitted stop/proceed decisions on dredging. Also, stations for monitoring air pollution were established. The Life Environment Division and the Agriculture and Fishery Division of Shizuoka Prefectural Government took the observations at these stations.

The observed items and the standards for action are given below. No abnormal values were observed.

Station	Observation Items	Frequency	Standard Values for Action
Basic	pH, DO, COD	1/day	COD not exceeding 8 ppm
	SS, Cl	"	PCB should not be detected
	transparency	"	Gross Hg 0.0005 ppm
	PCB, Hg	"	transparency 7 cm or more
Supplementary	transparency	4/day	7 cm or more
Outlet of drainage	PCB	1/day	0.01 ppm
	transparency	"	proportional to the above value
Underground Water	PCB	1/day	should not be detected
	H_2S	continuous	should not continue for 2 hours or more at 0.2 ppm at 2 points at the same time
Air	weather conditions	continuous	
	PCB	2/month	not exceeding 0.5 $\mu g/cm^3$
Fishes & Shellfish	PCB	1	not exceeding 3 ppm

CONCLUSION

The result of this sediment management effort was the test which, if successful, would permit the treatment of the remaining 370,000 m^3 of sediment. The excellent outcome, which created no secondary public nuisance factors, was a result of the refinement of techniques to prevent seawater pollution, H_2S gas and offensive odors.

If a large depot for temporary detention can be secured or sediments can be reclaimed on land, it is anticipated that expenses and quantities of cement and EGDP can be reduced.

REFERENCE

Ichikawa, Takashi. The Management of Sediments of the Port Tagonoura

THE IMPROVEMENT OF WATER QUALITY AT LAKE KASUMIGAURA BY THE DREDGING OF POLLUTED SEDIMENTS

Mineo Matsubara
Kasumigaura Work Office, Kanto Regional Construction Bureau
Ministry of Construction, Japan

LOCALE

LAKE KASUMIGAURA

Lake Kasumigaura is the second largest lake in Japan (Figure 1). It is located northeast of the Tokyo metropolitan area at the southeastern end of Ibaragi Prefecture. It actually consists of three lakes: Nishiura, Kitaura and Sotonasakaura. Nishiura Lake drains into Lake Sotonasakaura via the Kitatone River, and Kitaura Lake drains into Lake Sotonasakaura via the Wani River. Sotonasakaura Lake is connected with the Tone River via the Hitachi River at a point 18 km from the Pacific Ocean.

Lake Nishiura covers 171 km^2 and Lake Kitaura covers 34 km^2. The total area of Lake Kasumigaura, including Lake Sotonasakura and the rivers is about 220 km^2. The average depth of the lakes is 4 m. The deepest spot does not exceed more than 7 m. It is too shallow to maintain high water quality.

The average water level is approximately Y.P 1 m (the average sea water level + 16 cm). The lake volume is about 800 million. The lake used to be an inlet of the sea divided by the Kashima plateau. Then the mouth of the inlet was closed by sand and soil deposited by the Tone River. The water of the present lake therefore is a mixture of fresh water and sea water.

DRAINAGE BASIN

The area of the Kasumigaura basin is 2169 square kilometers - about 35% of Ibaragi Prefecture (Figure 2). The basin consists of the hilly country between 20 and 30 m above sea level and the rice fields extending along the coastal region of Lake Kasumigaura, except for parts of Mount Tsukuba (876 m elev.), Mount Ashio (628 m elev.) and Mount Kaba (709 m elev.) which are located along the reaches of the Sakura and Koise Rivers which flow to Lake Nishiura.

The average annual rainfall is 1350 mm. This produces an annual water input of 1.2×10^9 m^3 to Lake Kasumigaura. Dividing this volume by the approximate capacity of the lake yields an annual replacement rate of 1.5 times.

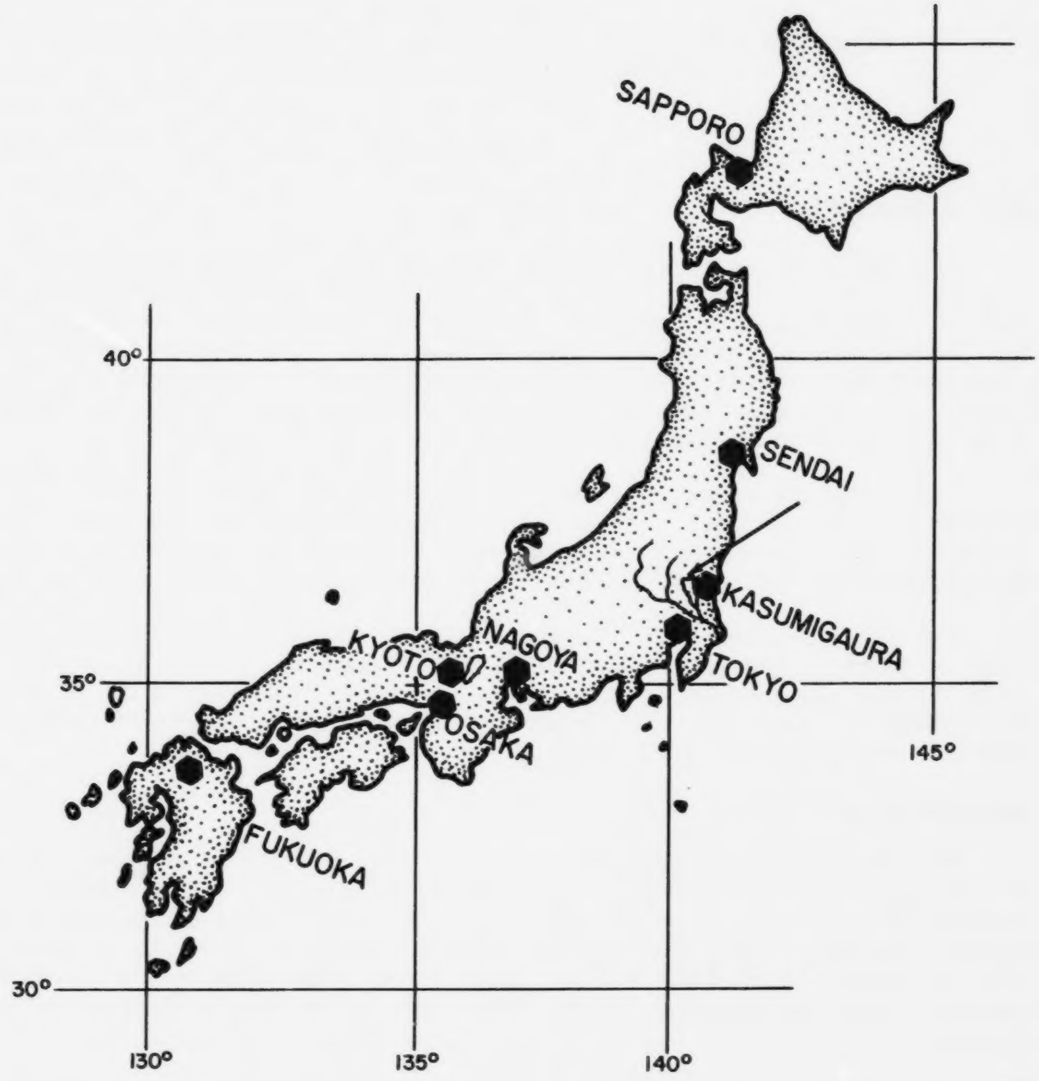

SAPPORO
SENDAI
KASUMIGAURA
TOKYO
NAGOYA
KYOTO
OSAKA
FUKUOKA
40°
35°
30°
130°
135°
140°
145°

Figure 1. The location of Lake Kasumigaura in Japan.

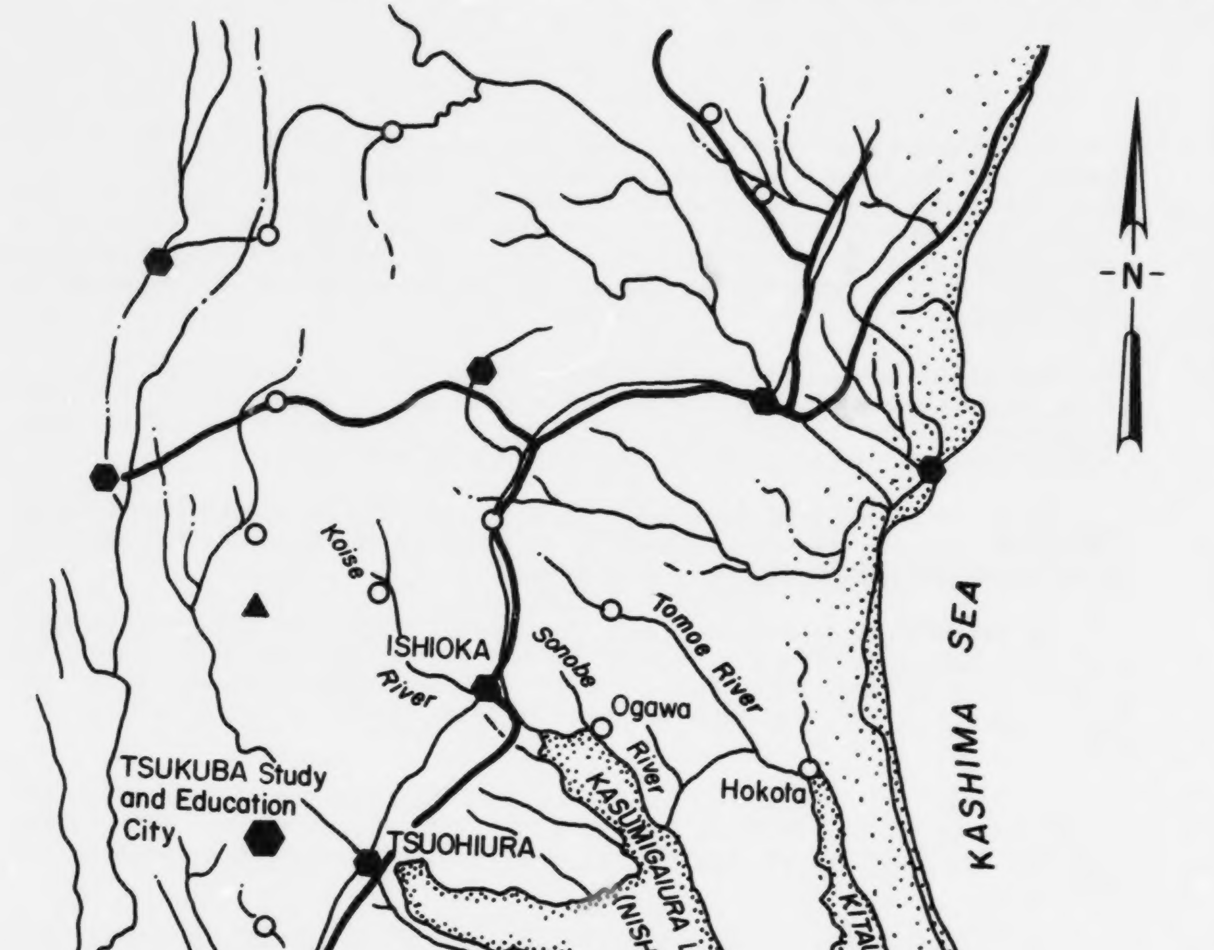

-N-
Koise
Tomoe River
KASHIMA SEA
ISHIOKA
Sonobe
River
River
Ogawa
River
TSUKUBA Study and Education City
KASUMIGAIURA LAKE (NISHIURAI)
Hokota
TSUOHIURA
KITAURA LAKE
Ono River
Kashima
Kashima Port
Hitachi River
Hatachi River Sluice Gate
Tone River
NARITA
International Airport
SAKURA
ASAHI
YOKAICHIBA
0 5 10 15 20 km

Figure 2. Lake Kasumigaura and its drainage basin.

More than fifty rivers including the Ono, the Sakura, the Koise, the Sonobe, and the Tomoe flow into Lake Kasumigaura, while only the Hitachi Tone River (the Kita Tone River and the Hitachi River are generally known as "the Hitachi Tone River") flows out of this lake. The Hitachi Tone River has a complicated hydraulic mechanism because the slope of the river bed is gentle. The water level is affected by that of the Tone River, which is influenced by tidal levels.

The basin area includes 47 cities, towns and villages in Ibaragi and Chiba Prefectures. The population of this area is estimated at 720,000. Agriculture is the biggest industry in the region. This is one of the most prominent regions in Japan for hog farming and carp rearing.

Large-scale projects undertaken in this area include the Kashima Coastal Industrial Zone in the eastern part of Kasumigaura and the Tsukuba Study and Educational Institution in the western region.

At Kasumigaura, the Kasumigaura Development Project is starting to cope with the increased demand for water by the metropolis. By 1983, when this project is completed, water resources of 40 m^3/S will be developed by utilizing the 2.6 x 10^8 m^3 available between the water levels of Y.P 1.30 m and Y.P 0 m.

The lake will become a freshwater lake by temporarily employing the Hitachi River Sluice Gate to prevent tidal intrusion upriver. This sluice gate, at the juncture of the Hitachi and the Tone Rivers, was completed in 1963.

Thus, since the lake has been resurrected as a storage reservoir, the maintenance of good water quality is very important.

CONSERVATION OF WATER QUALITY IN LAKE KASUMIGAURA

WATER QUALITY AT THE PRESENT TIME

The water quality standard for Lake Kasumigaura was established in November, 1972 as "type A-(c), COD 3.0 ppm, provisional standard type B, COD 5.0 ppm" for the lakes and marshes in the area of Kasumigaura, Kitaura (including the Wani River) and the Hitachi Tone River.

The average figures for 1977 water quality, measured at eight places in Kasumigaura, are given in Table 1. Observation points are indicated in Figure 3. The COD, which is the typical index for water quality of lakes and marshes, is between 6.0 ppm and 7.9 ppm. This is considerably greater than the 3.0 ppm which is the standard for Kasumigaura. (Figure 4)

T-N and T-P, which are indices of eutrophication, are 0.77 ppm to 1.25 ppm and 0.04 ppm to 0.08 ppm, respectively. Both of these figures exceed the general standards of 0.15 ppm T-N and 0.02 ppm T-P.

Past changes in water quality are not well known because the data are insufficient. However, a deterioration of the water quality has been conspic-

TABLE 1. AVERAGE LAKE KASAMIGAURA WATER QUALITY IN 1977

Basin	Observation Points	Chemical/Physical Variables					Number of Coliform Bacteria MPN/100 ml	Nutrients	
		pH	DO ppm	COD ppm	BOD ppm	SS ppm		T-N ppm	T-P ppm
Nishiura	Offshore at Kakeuma	8.0	9.6	7.6	3.3	22	120	1.25	0.06
	Offshore at Tamatsukuri	7.9	9.1	7.9	3.2	16	210	0.99	0.08
	Center of the lake	8.0	9.8	7.1	2.5	12	51	0.88	0.05
	Offshore at Aso	7.8	9.3	7.4	2.6	14	47	0.84	0.05
Kitaura	Offshore at Kamaya	7.8	9.9	6.0	2.1	7	290	0.80	0.04
	Jingu-Bashi	7.8	9.9	6.7	2.7	15	510	0.77	0.06
Hitachi Tone River	Sotonasakaura	7.8	9.8	7.3	2.5	20	190	0.85	0.05
	Ikisu	7.8	9.7	7.3	2.6	18	180	0.80	0.07

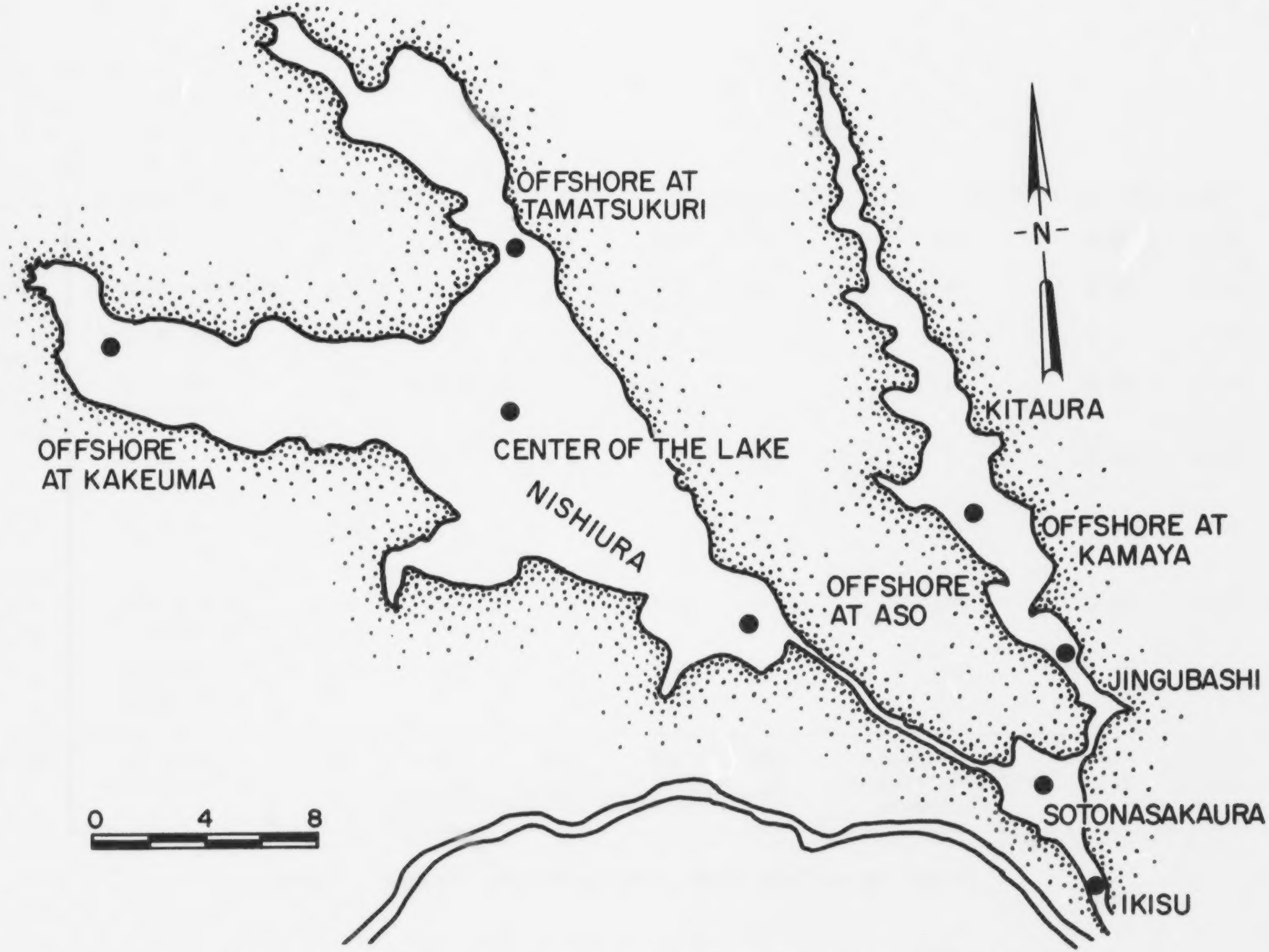

OFFSHORE AT
TAMATSUKURI
-N-
KITAURA
OFFSHORE
AT KAKEUMA
CENTER OF THE LAKE
NISHIURA
OFFSHORE AT
KAMAYA
OFFSHORE
AT ASO
JINGUBASHI
SOTONASAKAURA
IKISU
0
4
8

Figure 3. Observations points in Lake Kasumigaura (see Table 1).

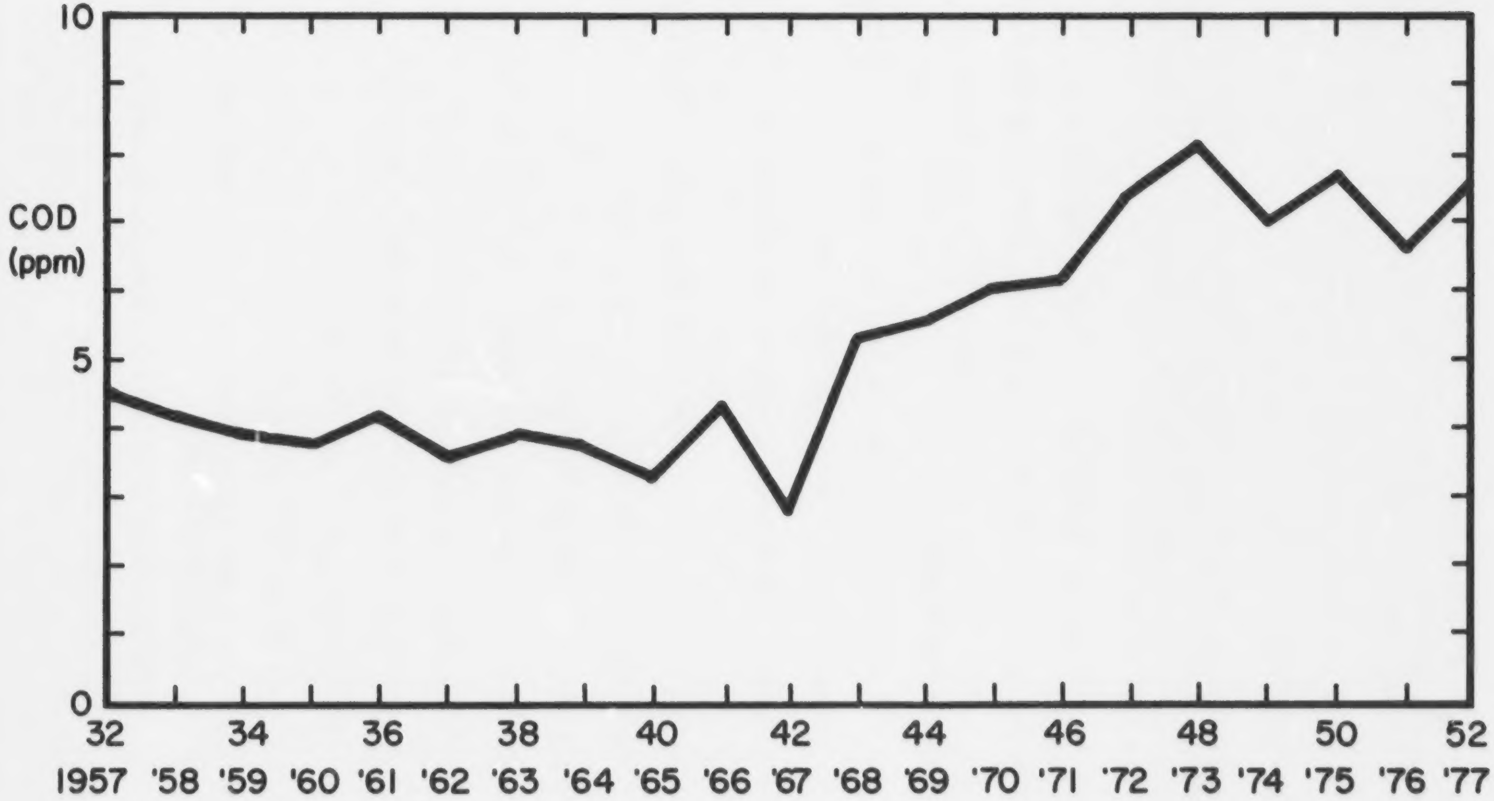


Figure 4. Change in COD (yearly average figures for the center of Lake Kasumigaura).

uous since 1968. Eutrophication of the lake has developed rapidly. In the summer, as water temperature increases, phytoplankton blooms are common. This algal bloom causes the surface of the lake to look as if green paint were floating on it. In the summer of 1973, a *Microcystis* bloom caused the drinking water to smell unpleasant and also killed many fish.

CONSERVATION MEASURES

The Japanese government and the prefectural authority are improving the water quality of the Kasumigaura basin. They are trying to restore it to conditions prevailing in 1965 and to set a standard for research and implementation of this type of project.

General techniques for conservation of water quality are:

(1) Removal of pollutants at the source by regulating drainage.

(2) Removal of pollutants from the river which feeds the lake by constructing a treatment pond.

(3) Removal of pollutants from the lake by dredging polluted sediments.

(4) Dilution of the polluted water with clean water.

The best technique or combination of techniques depends on the particular lake or marsh.

At Kasumigaura the following operations are planned or underway:

(1) Establishment of sewage and storm drainage.

(2) Disposal of livestock wastewater.

(3) Regulation of industrial drainage.

(4) Dredging of polluted sediments from feeder rivers.

(5) Dredging of polluted sediments from Lake Kasumigaura.

(6) Removal of *Microcystis aeruginosa*.

(7) Introduction of clean water to dilute the polluted water.

(8) Cleanup of Lake Kasumigaura by the inhabitants of the coastal cities, towns and villages.

DREDGING POLLUTED SEDIMENTS FROM LAKE KASUMIGAURA

Drainage from residential, industrial, and agricultural activities is the source of impurities found in the water of Lake Kasumigaura. The bottom sediment of the lake itself also releases impurities to the water. Therefore, dredging has been done since 1975 as a water quality improvement measure and is included in the overall project for cleaning up the river.

To eliminate the sediment as a pollution source to Lake Kasumigaura, 1.2 x 10^6 m^3 of sediment must be dredged. About 3.0 x 10^5 m^3 of the total are to be dredged by 1981 as part of the project for controlling the water sources of the Kasumigaura. This project in turn is part of the General Development Project for Kasumigaura.

A special pneumatic pump cutterless suction dredge, the "Kasumi", was developed for exclusive use on this project. It is designed to prevent excess turbidity and secondary pollution. The "Kasumi" and another dredge of the improved type, the "Koryu", have been dredging offshore from the mouth of the Sakura River at Tsuchiura City (Figure 5). About 4,000 m^3 were dredged in 1975, 18,000 m^3 in 1976, and 13,000 m^3 in 1977. The depth is between 1.4 and 5.7 m after dredging and the thickness of the cut is 0.5 m on the average. The water content averages 260% and ranges from 140% to 400%. Water content was computed according to the formula:

$$w = \frac{Ww}{Ws}$$

where: w = water content, dry wt. basis
Ww = weight of water
Ws = weight of solids

The dredged mud was pumped by pipeline to the disposal area two kilometers distant from the dredge, except in 1975 when the mud was transported by barge. The disposal area will become a park belonging to Tsuchiura City. When the mud at the disposal area is dry, is will be covered with soil and the facilities for the park will be constructed.

THE TWO DREDGES

The "Kasumi", a Sludge Dredge with Pot Type Suction Head and Pneumatic Pump, 60 m^3/H Negative Pressure Suction and Positive Pressure Discharge.

In 1971 the Sectional Committee for Studying the Development of New Types of Machinery and Methods of Reclamation began meeting. It was part of the Engineering Management Conference sponsored by the Bureau of Construction, Kanto District.

To establish a method for dredging the bottom while minimizing the resuspension of sediment, the dredge "Kasumi" was constructed. The following conditions were used to develop the dredge:

(1) The dredge should operate with a higher percentage of mud content than a hydraulic dredge.

(2) The turbidity caused by dredging should be curtailed as much as possible.

(3) The dredging cost should be less than the current costs.

These conditions for development led to the utilization of the "air-lift" (pneumatic) system developed in Italy. The method employs cylindrical pump bodies pushed into the bottom sediment. Modification of the pneuma system

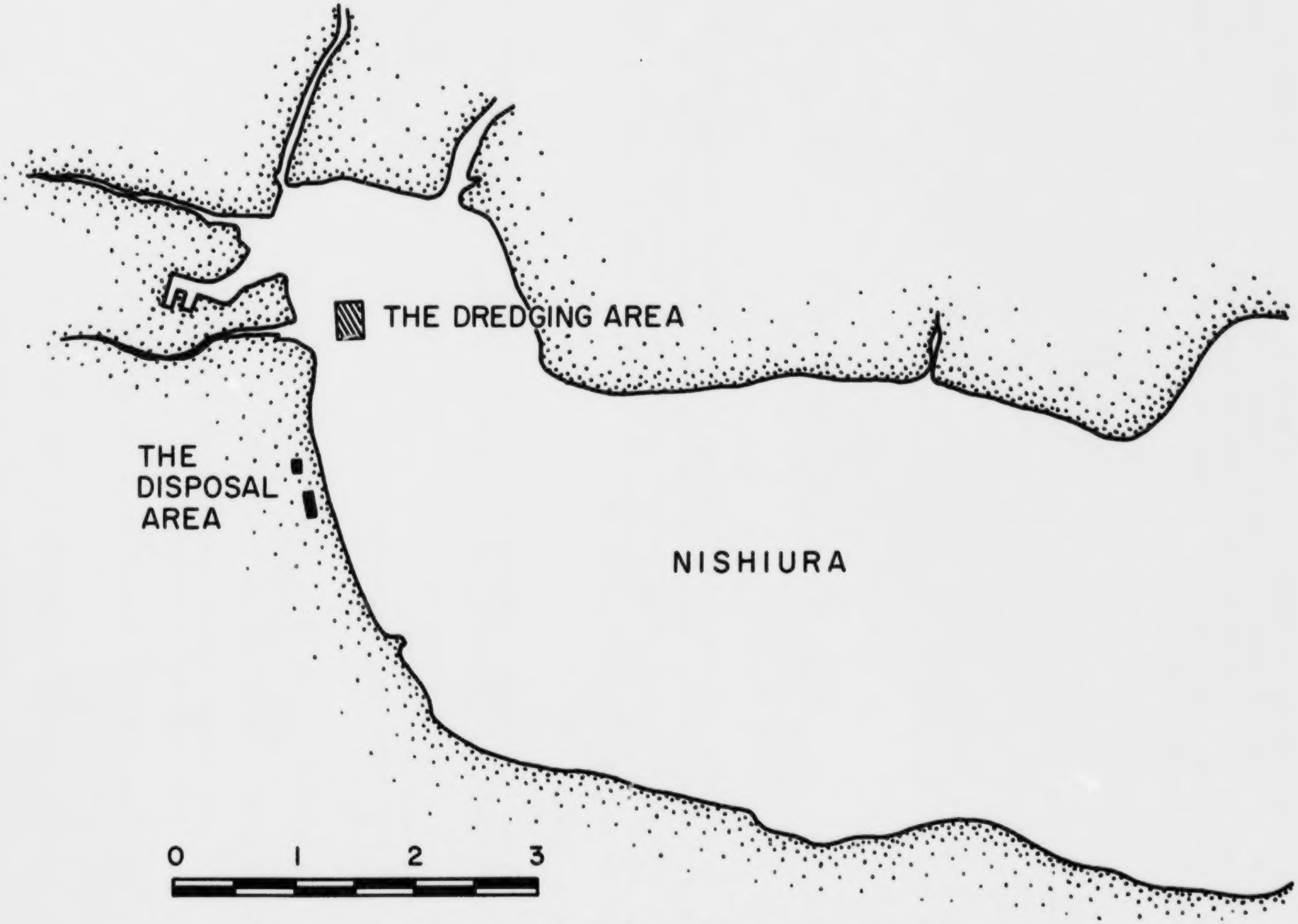

THE DREDGING AREA
THE DISPOSAL AREA
NISHIURA
0
1
2
3

Figure 5. Dredging and disposal area for bottom sediments.

employs a vacuum inside the pump body to suck bottom sediment in. Compressed air is then forced into the pump body to push the mud out (Figure 6). This dredge was constructed with the same specifications as the 200 ps class of pump dredge regarding draft, clearance and beam so that the clearance of bridges and the span of bridge piers will pose no problem when it is being transported. A plan of the sludge dredge is shown in Figure 7 and the specifications are given in Table 2.

TABLE 2. MAIN SPECIFICATIONS OF SLUDGE DREDGE "KASUMI"

Type: Pot type suction head. Negative Pressure Suction and Positive Pressure Discharge
Capacity: 60 m^3/H
Maximum Discharge Distance: 30 m
Maximum Depth of Dredging: 5 m
Dimensions of the Hull: 60 m x 5 m x 0.9 m (Length x Beam x Draft)
Weight: 78 Tons
Engine: Water-Cooled Type, 4 Cycles, Diesel Engine.
Output Horsepower: 220 PS
Generator: Waterproof, Self-Cooling, with Self-Excited Generator.
Output Power: Continuous Duty 180 KVA
Voltage: 440 V

The pontoons which provide flotation also house the generator, vacuum pump, air compressor and winch. The boom with the mud-sucking equipment is installed at the front of the dredge. A vacuum pump and air compressor are both used for dredging. The inside of the suction head is alternately pressurized and depressurized. An automatic valve causes the sediment to be sucked up and sent down the discharge pipe. The swing wires at the front and the anchor wires at the rear are used to move the suction head and are operated by a winch on the pontoon. Two suction heads are used to provide continuous cycle dredging. The volume of one cycle is 0.5 m^3 and the duration averages 60 seconds. Thus the dredging volume per hour is estimated at 60 m^3. The ship is designed with a maximum height of less than 2 m to clear overhead obstructions such as piers and bridges.

The discharge distance is only 30 m. This is because the mud is discharged onto a barge where the surface water drains into the river so that only the mud is transported to the disposal area.

The sediment near cities and towns is denser because the surface is covered at high water and uncovered at low water. This dredge was developed and manufactured to be used under these conditions. Its use in Lake Kasumigaura area required little alteration. Special features are:

(1) Pot type. Mud suction head with pneumatic pump.

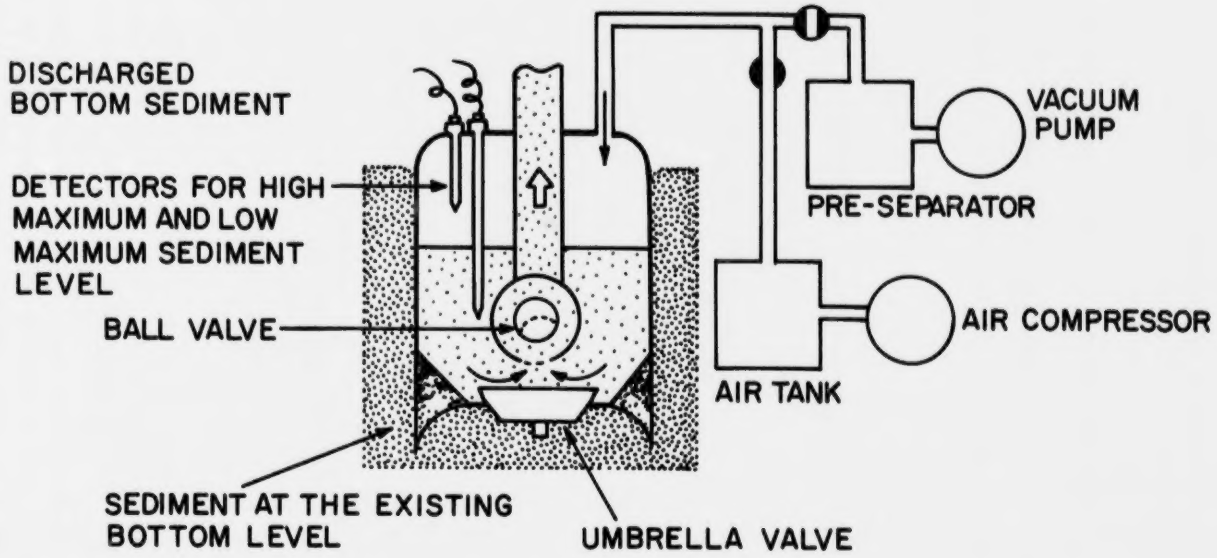

DISCHARGED
BOTTOM SEDIMENT
DETECTORS FOR HIGH MAXIMUM AND LOW MAXIMUM SEDIMENT LEVEL
BALL VALVE
SEDIMENT AT THE EXISTING BOTTOM LEVEL
UMBRELLA VALVE
AIR TANK
AIR COMPRESSOR
PRE-SEPARATOR
VACUUM PUMP

Figure 6. Dredging method employing negative pressure suction and positive pressure discharge.

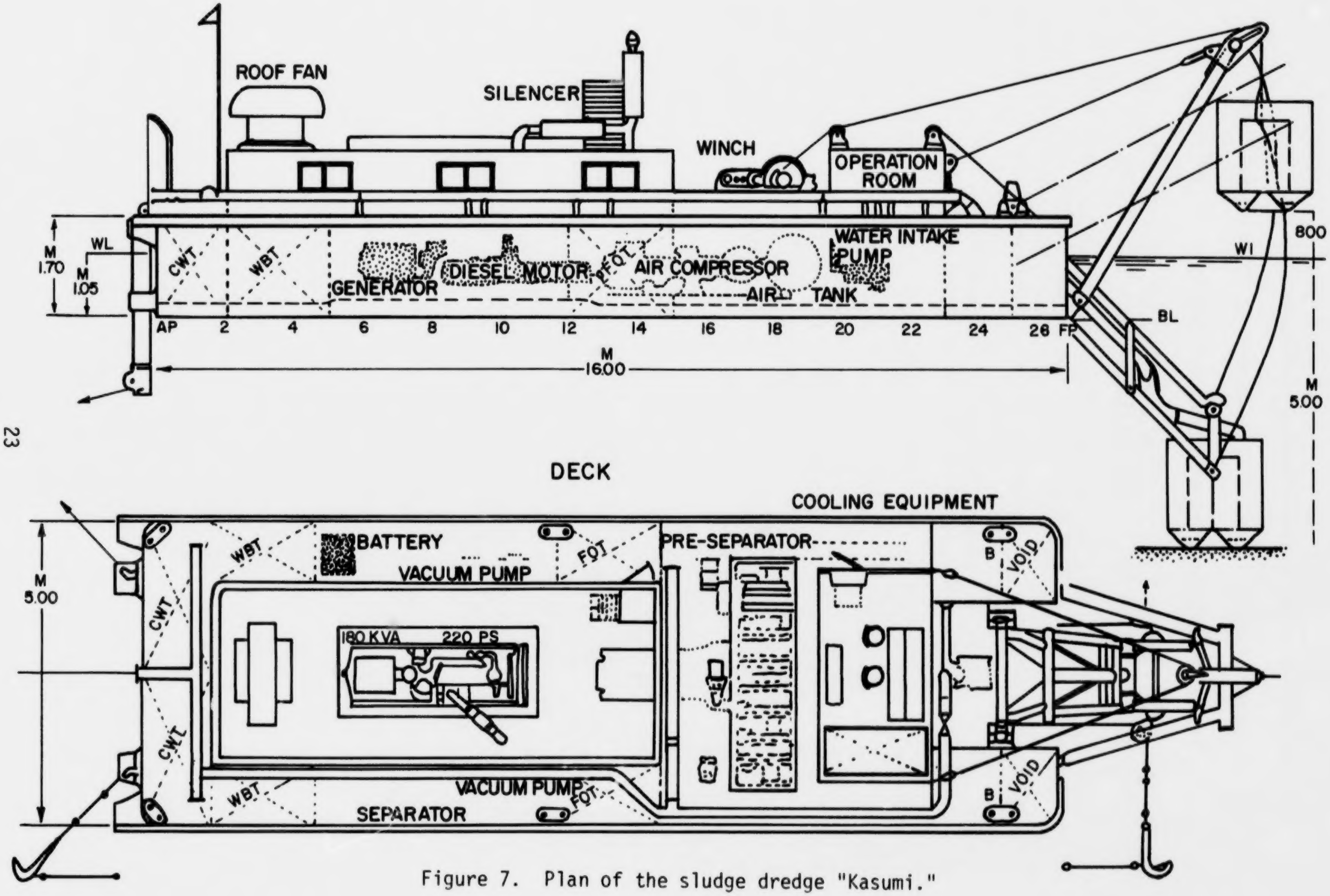

ROOF FAN
SILENCER
WINCH
OPERATION ROOM
WATER INTAKE PUMP
M 1.70
WL
M 1.05
CWT
WBT
GENERATOR
DIESEL MOTOR
F.O.T.
AIR COMPRESSOR
AIR TANK
800
WI
BL
AP
2
4
6
8
10
12
14
16
18
20
22
24
26
FP
M 16.00
M 5.00
DECK
COOLING EQUIPMENT
BATTERY
WBT
VACUUM PUMP
FOT
PRE-SEPARATOR
B
VOID
M 5.00
CWT
180 KVA
220 PS
CWT
WBT
VACUUM PUMP
FOT
SEPARATOR
B
VOID

Figure 7. Plan of the sludge dredge "Kasumi."

(2) Ability to dredge without causing turbidity.

(3) Disposal of water in the river instead of at the disposal area.

The results of the performance test are shown in Table 3.

The "Koryu", a Sludge Dredge with Swing Type Suction Head and Pneumatic Pump 100 m^3/H Negative Pressure Suction and Positive Pressure Discharge.

Dredging has been done in the Lake Kasumigaura region since 1975. The dredge "Kasumi" worked from 1975 to 1977. The volume dredged had been estimated at 1.2×10^6 m^3. About 3.0×10^5 m^3 of this was to be dredged by 1982, in accordance with the project plan regulated by the Special Law for Securing Water Sources. The dredge "Kasumi" had too small a capacity to fulfill this project and it could not dredge the lake bottom smooth; it left the bottom wavy. Therefore it was necessary to build another dredge.

When a sludge dredge is constructed today, environmental factors weigh heavily. A dredge of large capacity would not be favored over one of small capacity which caused less secondary pollution. The requirements for an environmentally sound dredge are that it dredge the lake bottom flat, that the suction head or swing does not generate excess turbidity, that the discharge has a high mud content and that it be capable of discharging the mud for a long distance through a pipeline. Two kinds of suction heads lend themselves to this kind of work. One is the pot type, the other the swing type (Figure 8). The "Kasumi" was a pot type. It was decided to build the new dredge as a swing type.

This office built the sludge dredge "Koryu" in 1977. It was paid for by the budget of the Construction Machinery Arrangement Expenses. It was designed as both an experimental and operational machine. The basic structure of the dredge "Kasumi", built in 1971, was adopted for use on the "Koryu". The new dredge was equipped with a swing type pneumatic pump suction head instead of the pot type.

The dredge was completed in March, 1978, and is shown in Figure 9 with specifications given in Table 4. Following are the improvements over the "Kasumi" design:

(1) Mud suction mouth of the swing pneumatic pump type.

(2) Mud collection equipment which does not generate turbidity. It has a drag suction head with grating.

(3) Sonic thickness monitoring equipment.

(4) Spuds, anchors and wires used to move the dredge.

(5) Long discharge distance.

(6) No overall turbidity.

(7) No disposal of effluents.

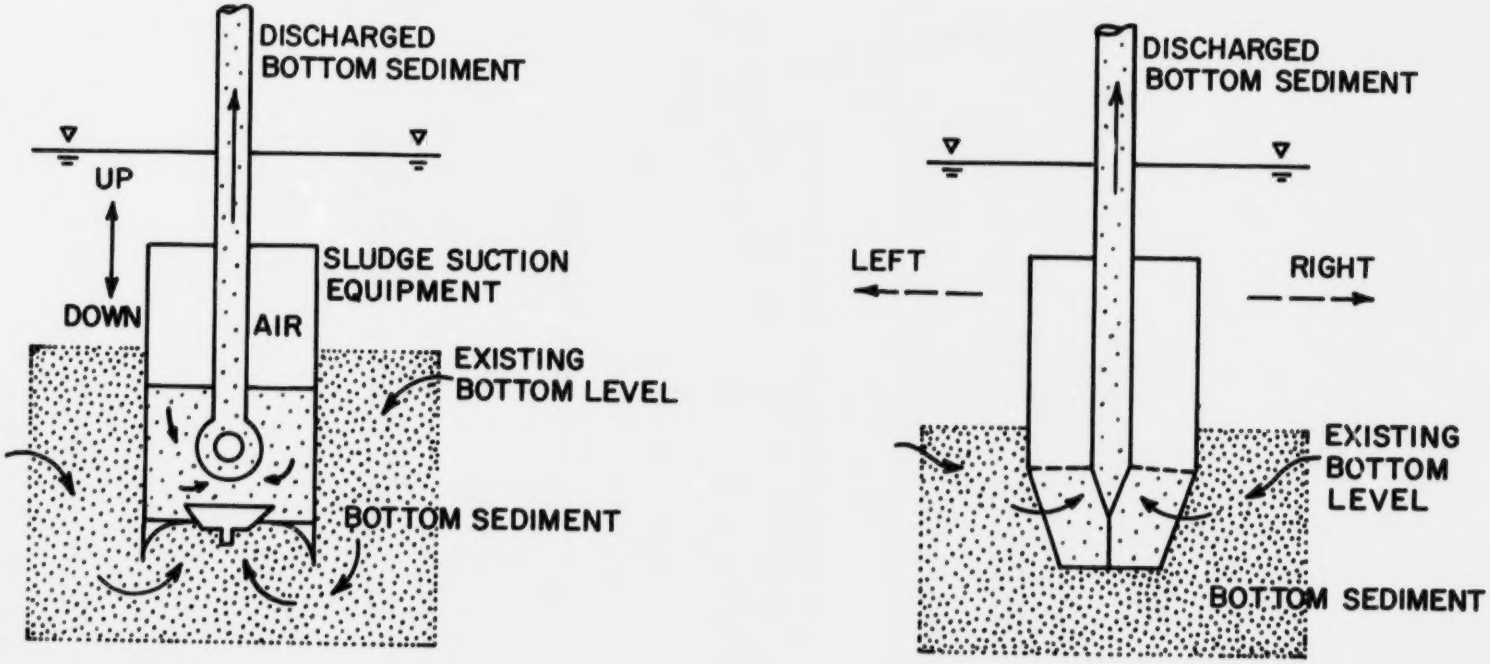
TABLE 3. PERFORMANCE TEST RESULTS

Test No.	River	Suction Head	Test Condition Water Depth (m)	Average Thickness of Sediment (m)	Discharge Length (m)	Discharge Height (m)	Duration of Test
1.	Tama				53	2.5	25'40"
2.	Tama	Pot-type	2	1.5	53	5.0	30'35"
3.	Tama	Pot-type	2	1.5	53	5.0	21'00"
4.	Tama	Pot and Swing	3	0.8	53	5.5	24'03"
5.	Tsurumi	Pot-type	1.7	1.5	18	5.0	24'16"
6.	Tsurumi	Pot-type	1.5	1.5	18	4.8	25'12"
7.	Tsurumi	Pot-type	2.0	1.0	18	5.3	27'18"
	Kasumigaura	Pot-type	4	1.0 - 0.5	30	-	-

(continued)

Table 3 (cont.)

Results of the Test									
				Cycle					
Test No.	River	Dredged Volume (m^3)	Number of Times	Sec.	Discharge Volume (m^3)	Work Volume Per Hr (m^3/hr)	Average Discharge Pressure (kg/cm^2)	Average Suction Pressure (mm/hg)	Remarks
1.	Tama	50	55	28	0.91	117	0.6	80	Sucking water only
2.	Tama	40	48	38.2	0.83	78	1.0	184	Percentage of Water content 300-350%
3.	Tama	25	35	36	0.83	83	1.2	210	"
4.	Tama	39	39	37	0.85	85	0.95	175	"
5.	Tsurumi	13	16	91	0.81	32	4.1	500	Percentage of Water Content 150%
6.	Tsurumi	15	18	84	0.81	34	4.1	500	"
7.	Tsurumi	17	21	78	0.82	39	4.4	550	"
	Kasumi-gaura	-	-	-	-	20	4	500	Percentage of Water Content 200-250%


DISCHARGED BOTTOM SEDIMENT
UP
DOWN
SLUDGE SUCTION EQUIPMENT
AIR
EXISTING BOTTOM LEVEL
BOTTOM SEDIMENT
DISCHARGED BOTTOM SEDIMENT
LEFT
RIGHT
EXISTING BOTTOM LEVEL
BOTTOM SEDIMENT

POT TYPE VALVE

SWING TYPE VALVE

Figure 8. Pot type and swing type pneumatic pump suction head.

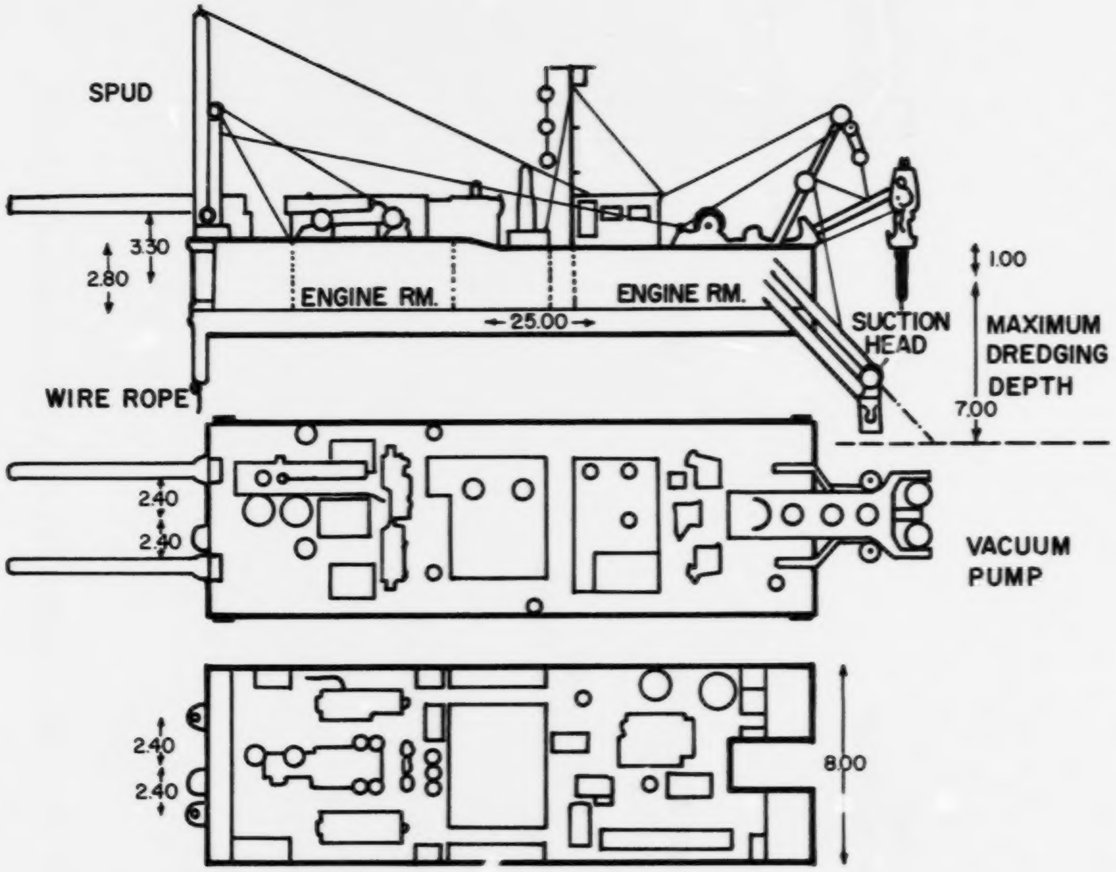

SPUD
3.30
2.80
ENGINE RM.
ENGINE RM.
25.00
SUCTION HEAD
1.00
MAXIMUM DREDGING DEPTH
7.00
WIRE ROPE
2.40
2.40
VACUUM PUMP
2.40
2.40
8.00

Figure 9. The sludge dredge "Koryu."

TABLE 4. MAIN SPECIFICATIONS OF SLUDGE DREDGE "KORYU"

Type: Swing Type Suction Head. Negative Pressure Sediment Suction and Positive Pressure Discharge
Capacity: 100 m^3/h (Water Content = 66%)
Maximum Discharging Distance: 2000 m
Maximum Dredging Depth: 7 m
Dimension of Hull: Length 25 m x Breadth 8 m x Depth 2.4 m x Draft 1.3 m
Weight: 260 Tons
Engine: Water Cooled, 4 Cycle, Diesel
Output Horsepower: 430 PS x 2
Generator: Waterproof, Self-Cooling, with Self-Exciting Generating Equipment
Output Power: Continuous Duty 350 KVA
Voltage: 440 V

To swing the dredge head an hydraulic winch winds anchor-fixed wire ropes through a sheave which is installed at the end of the dredging ladder. Spuds are put into the ground alternately to move the dredge while it is working. With the "Kasumi," the mud was sent to a dumping barge or discharged by a booster pump. In the new dredge, the sludge is first stirred in a stock tank installed in the dredge to change the viscosity. It is then sent to a secondary hopper. The dust is removed by a vibrating screen installed in the secondary hopper. The sediment is stirred again and discharged by pump. The volume of the flowing sediment can be controlled by a valve on the bypass pipe between the pump and the secondary hopper. Sonic thickness monitoring equipment is installed to check how much bottom sediment is dredged. Because of local conditions, the maximum height of the dredge while operating is less than 3.3 m. This dredge is appropriate for work at Kasumigaura because the water content of the shallow sediments is high.

The results of a short field performance test are given in Table 5. Although future actual daily operations will probably yield different results, it is clear that the designed nominal working capacity of 100 cubic meters per hour at 66% water content will be met. This also meets the planned annual dredging volume required by the program. From now on both dredges, "Kasumi" and "Koryu", will be used to dredge the bottom of Lake Kasumigaura.

SEDIMENT DISPOSAL STUDY

About 1.2 million m^3 of bottom sediment must be dredged by 1985. In Lake Kasumigaura, the sediments total 40-50 million cubic meters, according to rod surveys conducted at 35 points in the lake. The average thickness of the sediment is about 20 cm. These sediments had values of 11-27% ignition loss, indicating the presence of many organic substances. Therefore, it will be necessary to remove most of these sediments. It will be impossible to dispose of this large quantity of dredged sediment unless we make use of a special feature of the district. Although it's located on the fringe of the metropolitan area, 80% of the land (2200 km^2) is agricultural and the sediment can be disposed of on the farm lands.

TABLE 5. PERFORMANCE CAPACITY, SLUDGE DREDGE "KORYU"

No.	Discharge Volume (m^3/hr)	Water Content (%)
1	128	
2	110	
3	107	
4	106	
5	108	
6	102	
7	101	
8	97	
9	101	
	97 - 128	200 - 300%

Tsuchiura City (population 100,000) and Ishioka City (population 40,000) are located along the Jooban Line of the Japan National Railway at the northwestern part of the lake. The Kashima Coastal Industrial Zone is located downstream of the rivers, where the existing rice fields are to be reclaimed for the city.

Disposal techniques are under study to use the sediments for rice production and landfill for the city. Experiments are being conducted to compare rice growth in two kinds of soil, one comprised of sediment, the other the usual soil. Also underway are experiments to dry the sediment naturally on land surrounded by a soil embankment where the dredged mud is deposited to a thickness of 50-150 cm. The water in the mud separates as the sediment settles and consolidates. The water is drained and then permeates underground. The mud left behind dries in the sun. Low water content mud is used for this purpose. Following is the result of a survey from February, 1977 to March, 1978. The accumulated thickness of mud was 150 cm and the initial water content was 60-75%. The final water content was 54-60%. The complicated influences of weather, water table, and evaporation from the leaves of planted vegetation on the process of settling and consolidation will be studied. Officials plan to investigate ways to shorten the disposal period by using supplementary solidification chemicals.

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LAKE SUWA WATER POLLUTION CONTROL PROJECTS

Akira Sakakibara, Director
Osamu Hayashi, Chief, River Development Section
Civil Engineering Management Office for Lake Suwa and Surrounding Region
Nagano Prefecture Department of Civil Engineering, Japan

INTRODUCTION

More than a decade has passed since water pollution in Lake Suwa first attracted public attention. During this time, eutrophication of the lake has continued with no significant improvement in water quality. Algae continue to bloom from summer to fall every year.

In view of the seriousness of the problem, concerned public and private interests are intensifying their studies to develop drastic measures to purify the lake water. This is the tenth year of dredging. Partial use of the basin-wide sewage system will start next year. This paper outlines their efforts and other research and projects which are one part of the slow but steady progress being made to improve the water quality of Lake Suwa.

The water pollution control measures being taken in Japanese rivers and lakes were outlined last year at the Third U.S.-Japan Experts' Meeting on "Management of Bottom Sediments Containing Toxic Substances" (1). In Kato's paper (1) the dredging of bottom sludge at Lake Suwa, considered the most polluted lake in the country, is used to illustrate lake clean-up efforts in Japan.

BRIEF DESCRIPTION OF THE LAKE SUWA BASIN

Lake Suwa is located in the center of Nagano Prefecture at what is called the "roof" of the Japanese Archipelago. There are 31 rivers flowing into it, but only the Tenryu river flows out. The drainage area is 531 km^2, the height above sea level, 759 m, the planned lake area, 13.3 km^2, the circumference, 16.2 km, the maximum depth, 6.8 m, and the average depth, 4.0 m. It is also a major recreational fishery and water resource.

The four cities of Okaya, Suwa, Shimosuwa and Chino are on the periphery of the lake. These cities (combined population about 170,000) form an important industrial area producing optical and precision machinery and food products. In recent years, the modernization of lifestyles and expanding industry have led to water pollution in the lake. The eutrophication of Lake Suwa currently has reached its peak, with "Aoko" (Microcystis) growing in profusion in the summer, rapidly covering the lake like a green carpet. To make matters worse, Aoko creates an offensive odor as it dies and decomposes. The concomitant "Susumizu phenomenon" (oxygen depletion) adversely affects the lake's fishery. These factors contribute to the decreasing use of shoreline facilities by tourists and local residents.

Today, the dredging of the polluted sediments and the construction of the basin-wide Lake Suwa sewage system and related public sewage networks are being carried out to improve the environment of the four peripheral cities and prevent future contamination of the lake.

WATER POLLUTION OF LAKE SUWA

The periphery of Lake Suwa has been rapidly developing as an industrial area and tourist resort. The pollution of the lake is a by-product of the regional development.

Lake Suwa has long been known as a lake with extensive eutrophication but recent levels have been extreme. The inadequate sewage system around the lake is a significant cause of pollution, but the sludge which has accumulated on the bottom for a long time is also a contributing factor. The dominating impact of the contamination, from whatever source, is the nuisance growth of Aoko and clouds of mud in the water.

EXTERNAL FACTORS

The rivers and waterways flowing into Lake Suwa carry wastes from industry, homes, agriculture and hot springs. These loads create a BOD amounting to 14,000 kg per day (Table 1).

TABLE 1. EXOGENOUS BOD LOADS PER DAY (1972), LAKE SUWA

Classification	BOD
Industrial waste	10,634 kg
Domestic sewage	3,395 kg
Livestock excretions	209 kg
Raw sewage	137 kg
Agricultural waste	-
Total	14,375 kg

The advanced stage of eutrophication in the lake is attributed to nutrients such as nitrogen and phosphorus, which flow in via the river (Table 2).

INTERNAL FACTORS

Nitrogen and phosphorus input to the lake and, accompanied by high BOD, cause an abnormally fast growth of plankton and aquatic plants. These organisms in turn provide sources of nitrogen and phosphorus for other organisms, or die and become bottom sediments. The contamination of the lake is accelerated by this cyclic pattern.

TABLE 2. SOURCES OF NUTRIENT INPUT, LAKE SUWA

	Inflow			
	N load		P load	
Human wastes	420	(12.6)	116	(31.1)
Septic tanks	310	(9.3)	21	(5.6)
Sewage disposal facilities	546	(16.4)	47	(12.6)
Industrial wastes	363.8	(10.9)	53	(14.2)
Agricultural fertilizers	754	(22.7)	85	(22.8)
Mountains and forests	247.3	(7.4)	4.2	(1.1)
Fishery (feed)	99.8	(3.0)	8.5	(2.3)
Livestock wastes	55.6	(1.7)	24.6	(6.6)
Hot spring wastes	41.3	(1.2)	5.3	(1.4)
Gas well wastes	463.9	(13.9)	7.5	(2.0)
Rainwater	25.7	(0.9)	0.68	(0.2)
Total	3,327.3 (kg/day)	(100) (%)	372.7 (kg/day)	(100) (%)

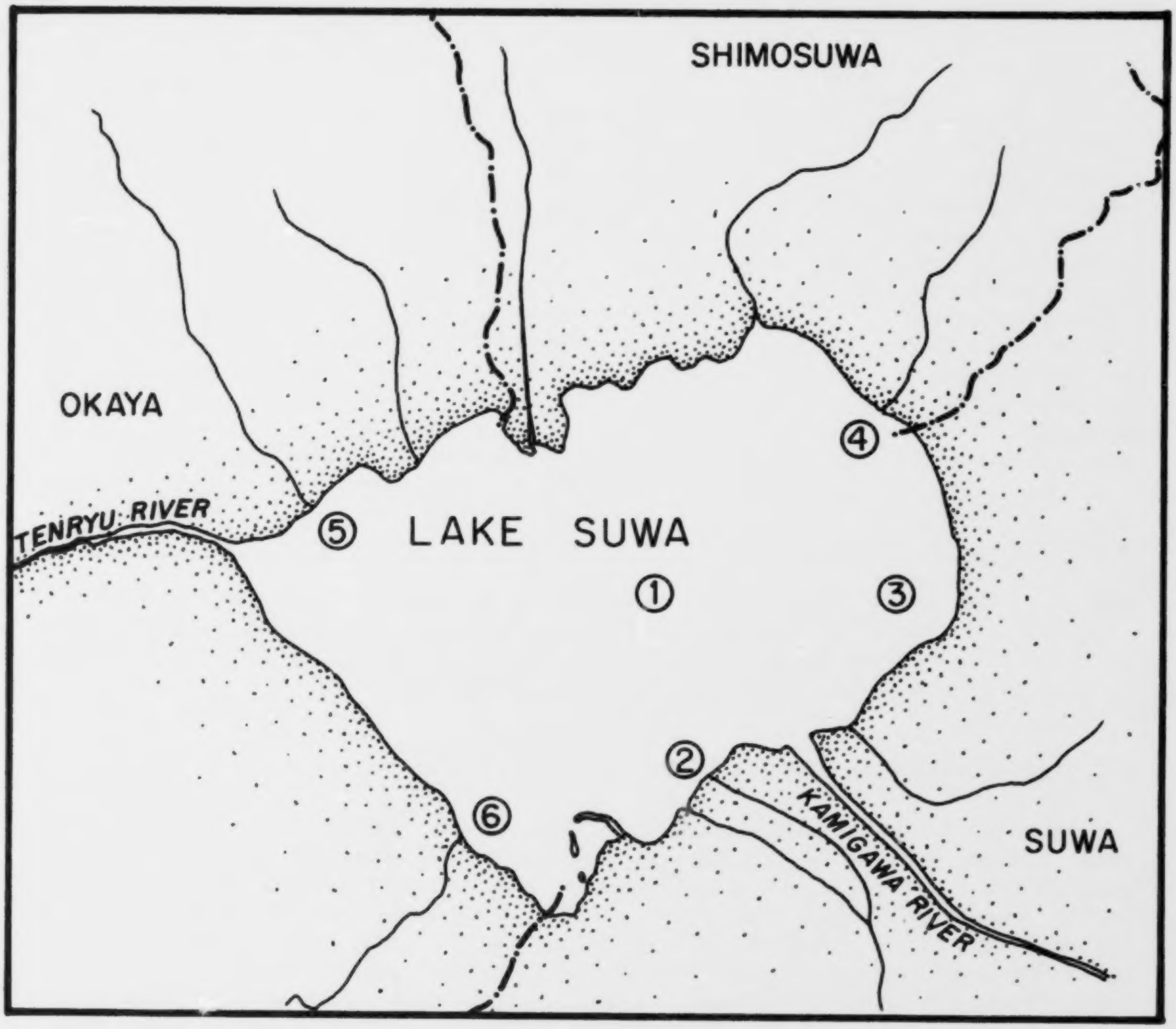
Note: The calculation of the above table is based on a run-off coefficient of 70%. The pollution unit employed is borrowed from the 1970 survey of the Ministry of Construction (Example: Human wastes N = 3.0 g/man-day, P = 0.83 g/man-day).

QUALITATIVE CHANGE IN WATER QUALITY

Studies of Lake Suwa have been conducted by various investigators since 1910. These provide some data on the historical changes in water quality. Because of differences in sampling and testing methods they may not be directly comparable, but they show trends of change to the present state. Figure 1 shows the location of sampling sites in Lake Suwa. Figure 2 represents the qualitative change in the lake's water quality based on such past data. Figure 3 shows the change in COD value in the center of the lake during the past six years. Figure 4 shows the water quality from April to December, 1977. The extremely high COD and SS values in July through September indicate the vigorous growth of Aoko (Microcystis). Line "A" shows the COD values of the lake water when filtered through filter paper (equivalent to JIS Class 5C: 0.45 μ pore size). The COD values are almost as low as the environmental standard of 3 ppm. This demonstrates that the water quality can be drastically improved by removal of the suspended matter, including Microystis.

Figure 5 shows the 1976 investigation of the influences of Aoko and other phytoplankton on the water quality at four points in the lake. In the figure, "upper layer" refers to a position 0.5 m below the surface, and "lower layer" 1.0 m above the bottom. The figure shows that the average COD values of the water after filtering out the Aoko are very close to the environmental standard. These data were collected in 1976 by the Suwa Health Center in accord-

(text continues on page 41)


SHIMOSUWA
OKAYA
TENRYU RIVER
⑤
LAKE SUWA
①
③
④
②
⑥
KAMIGAWA RIVER
SUWA

1. Center of Lake Suwa
2. 200 m offshore of the mouth of the Miyagawa river
3. West of Hatsushima Island
4. 200 m offshore of the mouth of Osawa river
5. 200 m offshore of the mouth of the Tsukama river
6. 200 m offshore of the mouth of the Obori river

Figure 1. Sampling sites in Lake Suwa.

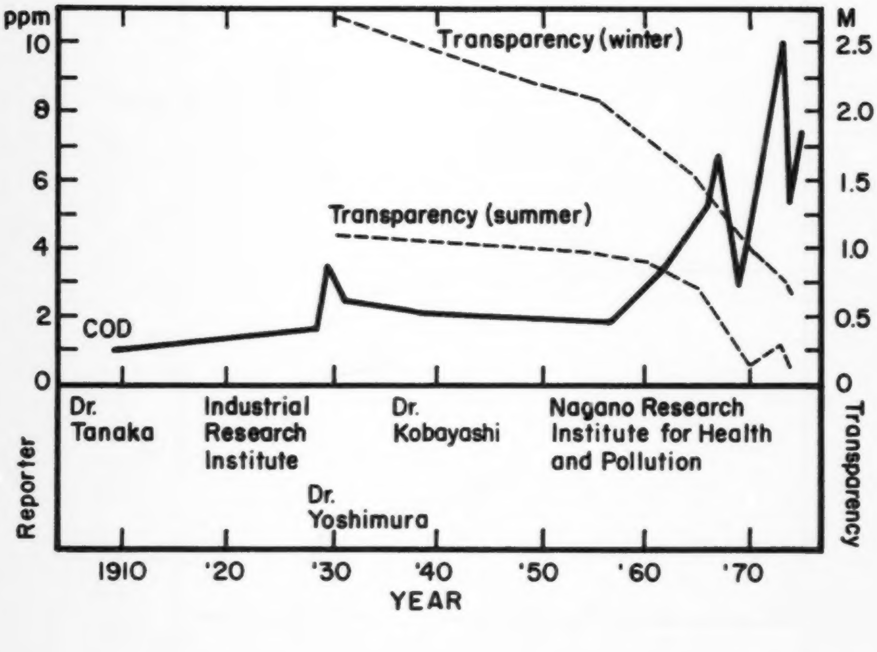


Figure 2. Qualitative change in water quality of Lake Suwa (2).

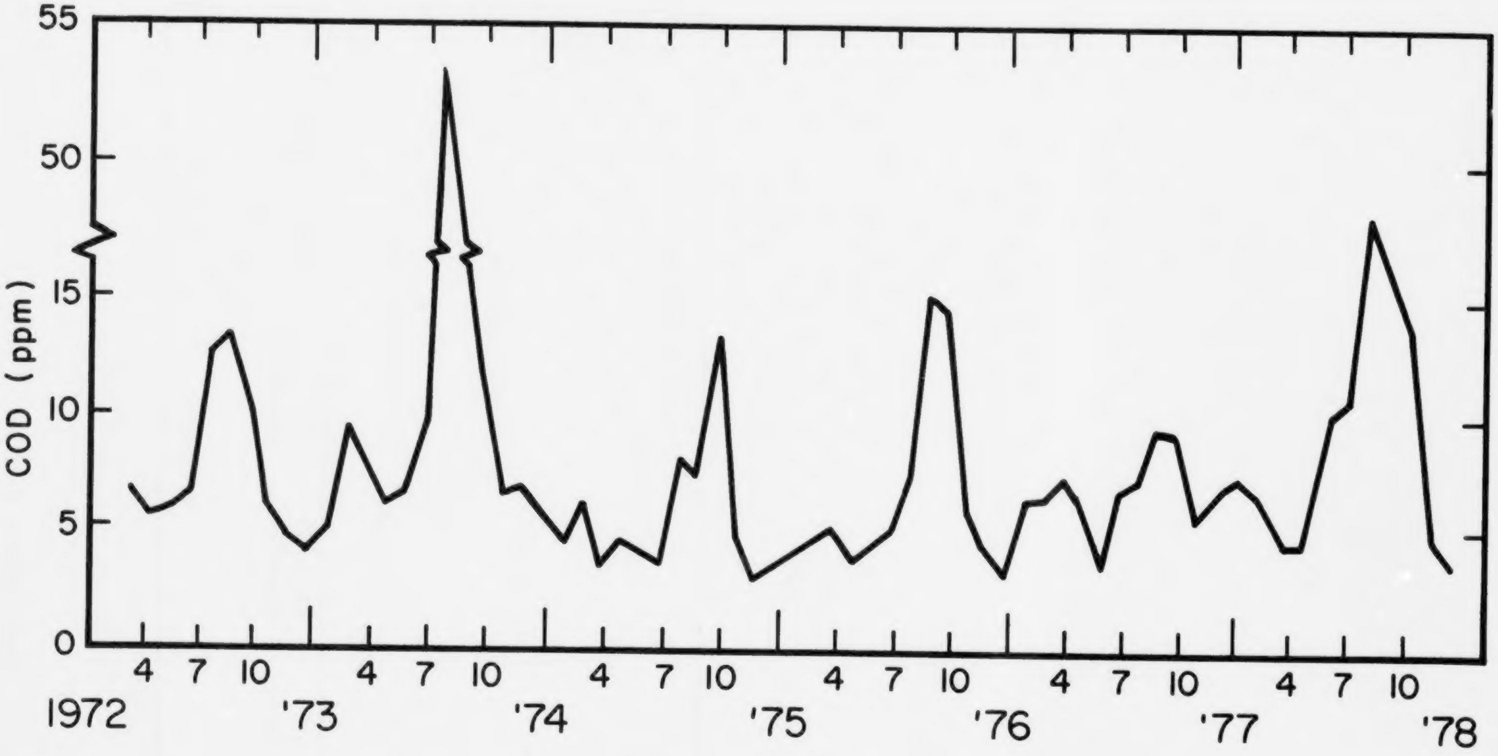
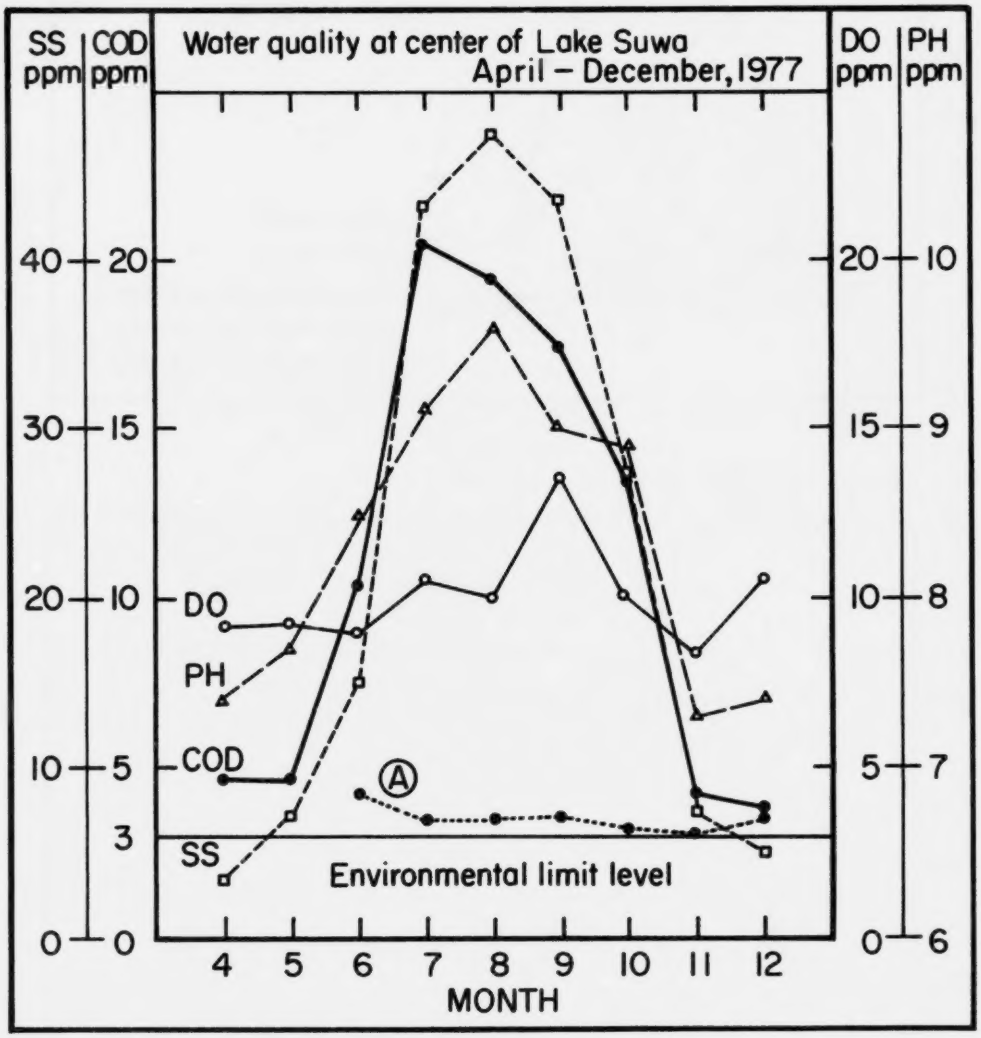


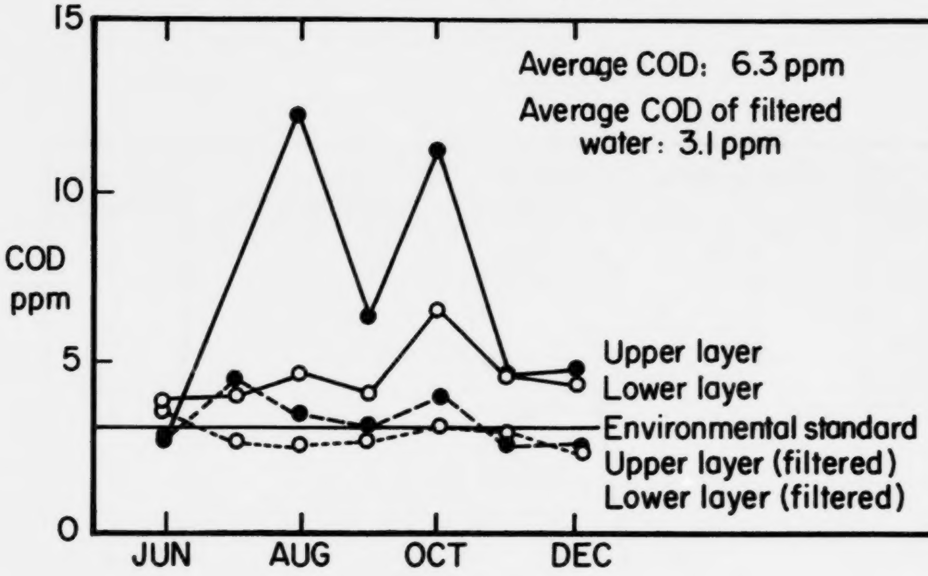
Figure 3. Change in water quality in Lake Suwa (lake center).



Note: Ⓐ represents the COD values of filtered lake water (Number 5C filter paper).

Figure 4.

200m offshore of mouth of the Miyagawa River



Center of Lake Suwa

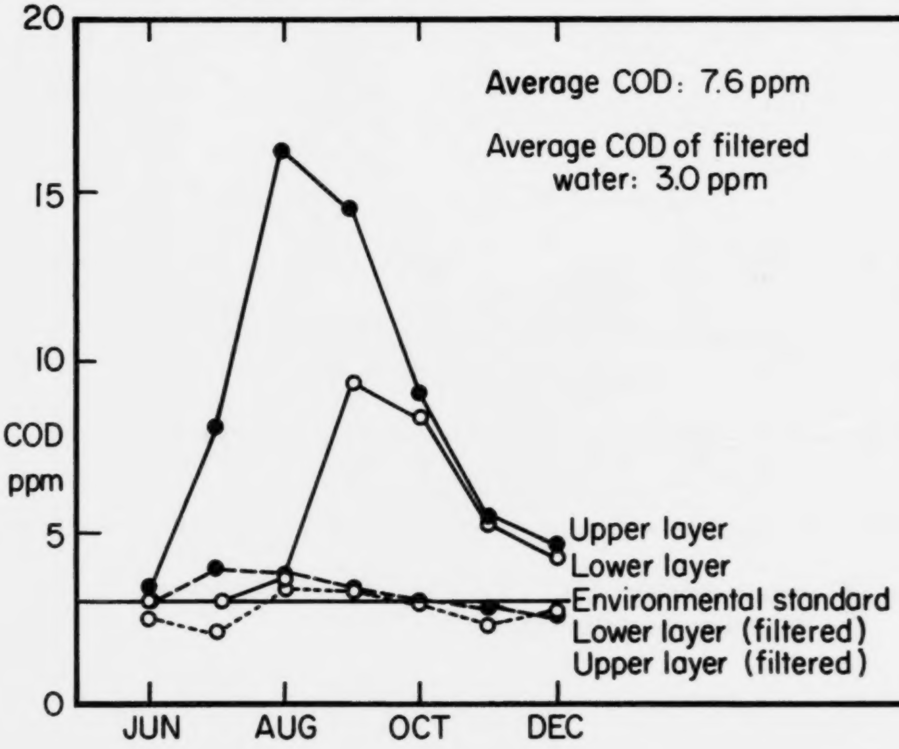


Figure 5. Effects of phytoplankton on lake water.

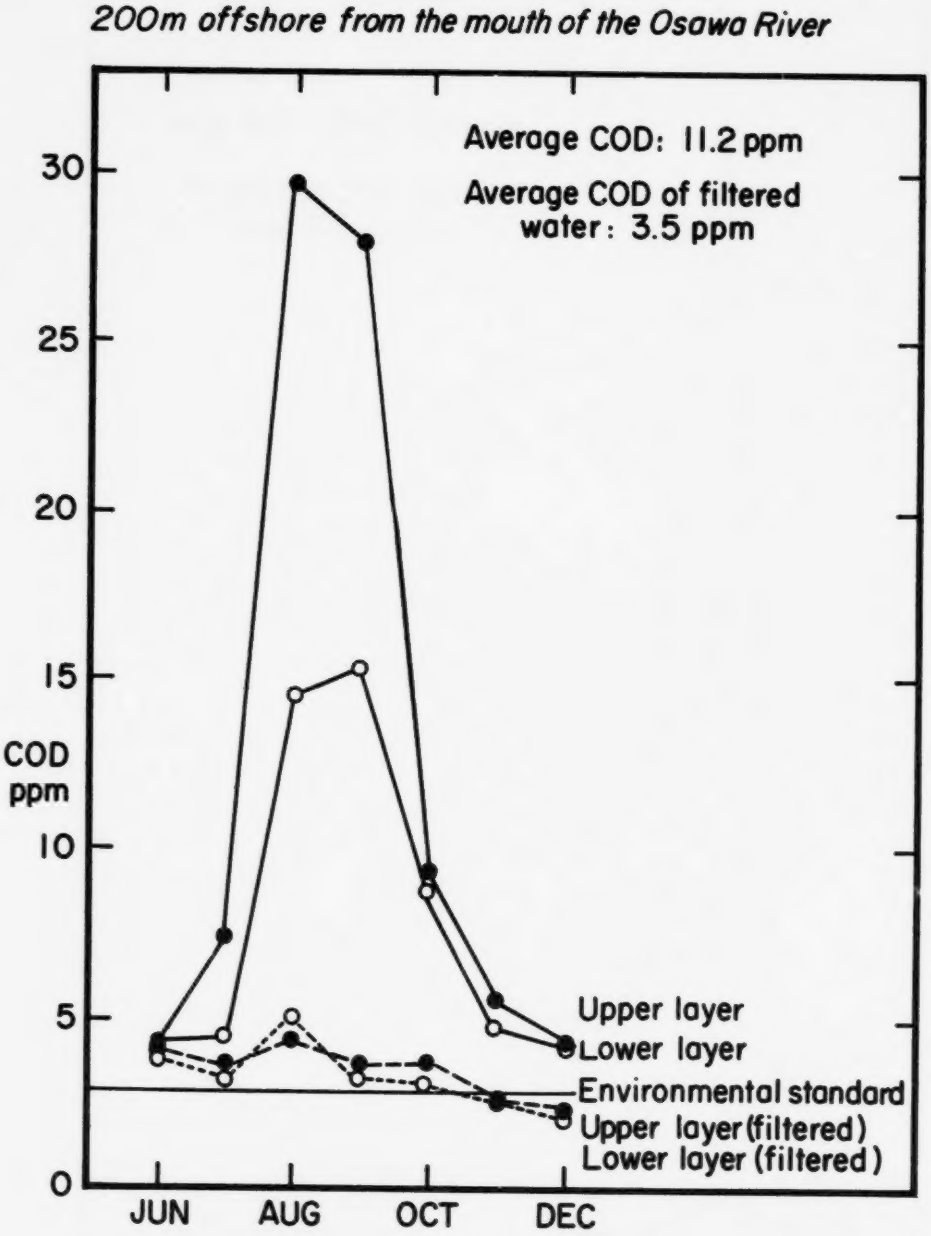

200m offshore from the mouth of the Osawa River
Average COD: 11.2 ppm
Average COD of filtered water: 3.5 ppm
COD ppm
30
25
20
15
10
5
0
JUN
AUG
OCT
DEC
Upper layer
Lower layer
Environmental standard
Upper layer (filtered)
Lower layer (filtered)

Figure 5.

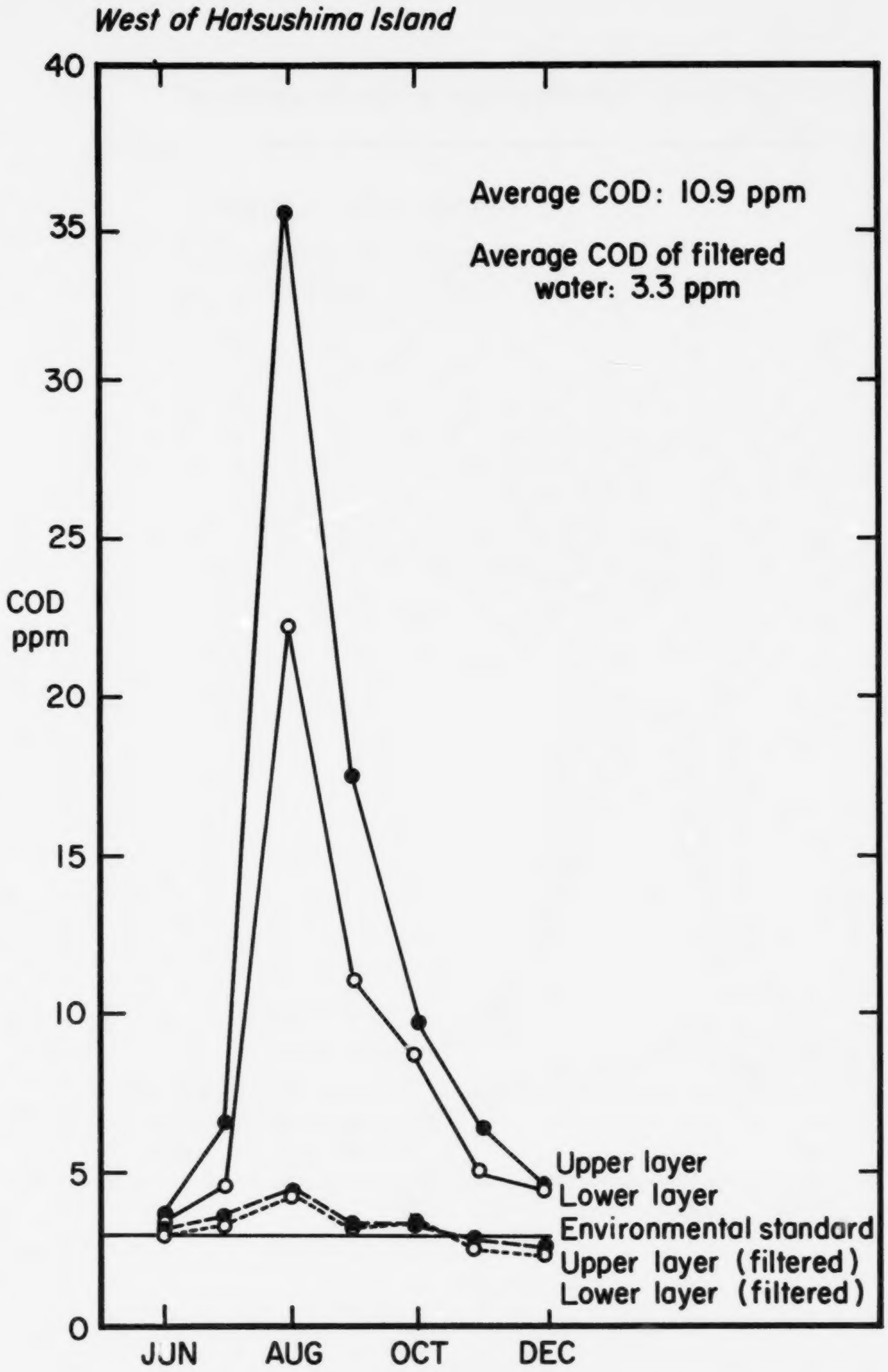

West of Hatsushima Island
Average COD: 10.9 ppm
Average COD of filtered water: 3.3 ppm
COD ppm
40
35
30
25
20
15
10
5
0
JUN
AUG
OCT
DEC
Upper layer
Lower layer
Environmental standard
Upper layer (filtered)
Lower layer (filtered)

Figure 5.

ance with the lake water measurement plan, 6-12/76. Filtering was done with 0.45 μ filter paper.

Figure 6 shows the results of water analysis at the center of the lake in an undredged area and at the west of Hatsushima Island where dredging has been carried out. These results show that dredging has little effect in lowering the DO value. Data in Figure 6 were taken from measurements of local water for public use (Pollution Section, Nagano Prefecture Department of Living Environment).

BOTTOM MATERIAL

Lake Suwa was divided into 13 1-km square grids. The bottom material in each of the 13 grids was analyzed in 1976 and 1977. Figure 7 shows the average results of the analyses.

Currently, there seems to be no explicit standard for the removal of organic sludge. However, elsewhere in the country, the following criteria have been used:

Ignition loss	15% or more
COD	20 mg/g or more
Sulfides	1.0 mg/g or more

The above criteria can be used to conclude that there are about 60 cm of bottom sediment in Lake Suwa.

QUALITATIVE CHANGES IN ALGAL GROWTH

The most obvious impact of the eutrophication of Lake Suwa is the rampant growth of algae. The pollution of the lake may be characterized by the propagation of Aoko (Microcystis) during the summer time.

Diatoms (Diatomaceae) which generally grow in less eutrophic lakes were reported in 1918. By 1948 diatoms had increased in kinds and green algae and blue-green algae (Cyanophyta) were also found. In the 1960s the growth of Aoko began to attract attention. In the 1970s, its growth increased in intensity.

WATER POLLUTION CONTROL IN LAKE SUWA

The Lake Suwa Pollution Control Measures Investigation Committee was established in 1965 to study the lake's pollution and immediately took action to control and eliminate the pollution. The committee conducted investigations for two years and reported its conclusions in 1968. To control external sources of pollution it recommended the construction of the Lake Suwa Basin sewage treatment system to eliminate the influx of polluted water (Figure 8). To control the endogeneous sources the Committee suggested dredging the organic sludge deposited on the bottom.

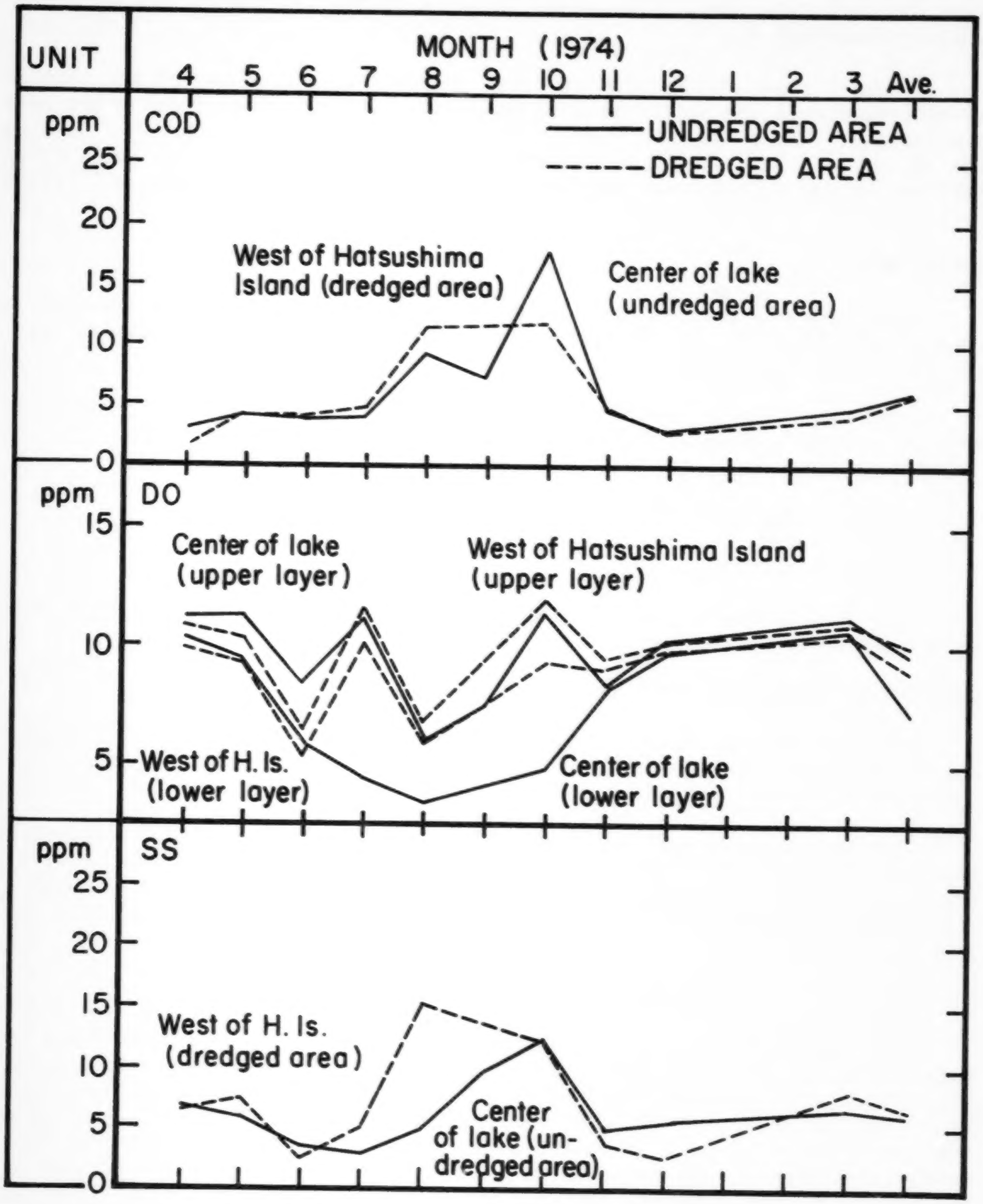


Figure 6. Results of water analysis at center of lake in an undredged area.

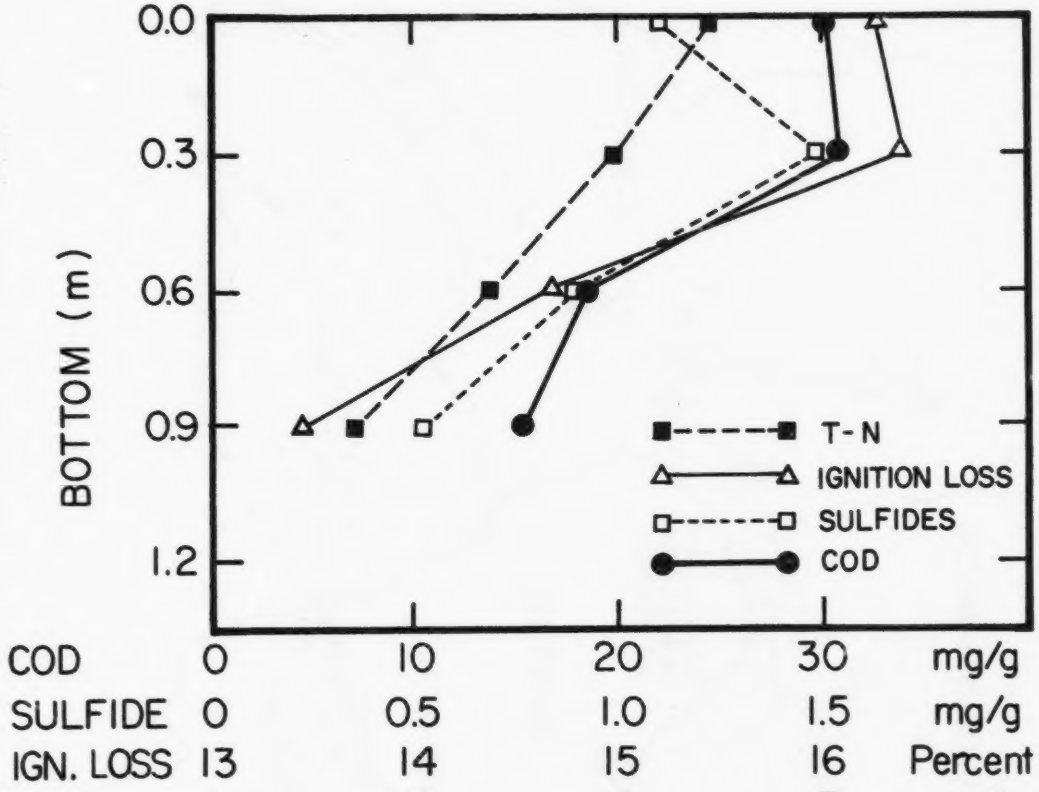


Figure 7. Average results of bottom sediment analysis in 1976 and 1977.

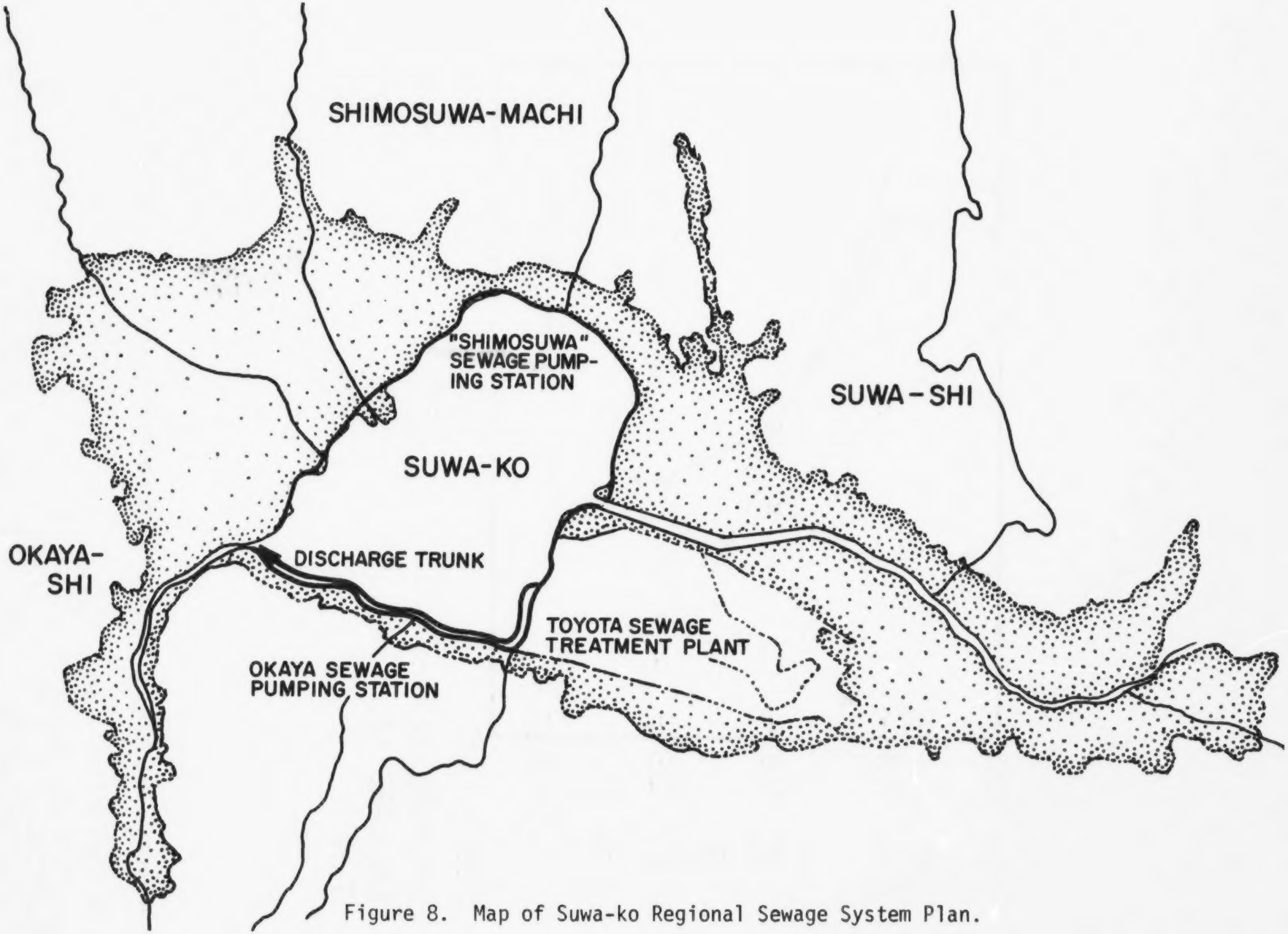

SHIMOSUWA-MACHI
"SHIMOSUWA" SEWAGE PUMPING STATION
SUWA-SHI
SUWA-KO
OKAYA-SHI
DISCHARGE TRUNK
TOYOTA SEWAGE TREATMENT PLANT
OKAYA SEWAGE PUMPING STATION

Figure 8. Map of Suwa-ko Regional Sewage System Plan.

LAKE SUWA BASIN SEWAGE SYSTEM CONSTRUCTION PROJECT

A major cause of pollution in Lake Suwa was raw sewage, so construction of a large-scale sewage and treatment system was a fundamental step.

The construction project began in 1971 and will be completed in 1992. Tables 3, 4, and 5 provide statistical data on the project.

TABLE 3. LAKE SUWA REGIONAL SEWAGE SYSTEM PLAN

District	Drainage Area (ha)	Future Population (Est. 1,000)	Total Distance (km)	Sewage Volume (1,000 m^3/day)	Treatment Process
Okaya	1,508		Koshu TS (Trunk Sewer) Chino TS		Ordinary Activated Sludge Process
Suwa	1,423				
Chino	715	1985: 200	Chuoh TS	1985: 334	
Shimosuwa	516	1990: 212	Nishiyama TS	1990: 337	
Total	4,162		25.6		

Table 3 (cont.)

Sewer System	Treatment Plant (GA ha)	Pumping Station (GA m^2)	Construction Period (FY)	Authority	Administrative Authorization
Separate System	Toyota Sewage Treatment Plant 18.7	Shimosuwa Sewage Pumping Station 1,320	1971 - 1985	Nagano Prefecture	Authorized by Prefecture Dec. 27, 1971 Authorized by City Planning Law March 24, 1972 Authorized by Sewage Law Feb. 28, 1972

GA = ground area FY = Fiscal year

(continued)

Table 3 (cont.)

Estimated Sewage Volume (m^3/day)

Item		Average Per Day	Maximum Per Day	Maximum Per Hour
Sanitary Sewage		76,337	95,425	143,140
Industrial Waste		220,310	220,310	330,468
Hot Spring Waste		17,257	21,571	32,357
Ground Water		23,395	23,395	23,395
Total	m^3/day	337,099	360,401	528,954
	m^3/hour	14,000	15,000	22,000
	m^3/sec	3.9	4.2	6.1

Outline of Sewage System

Item		Pipe Diameter (ø m)	Distance (km)
Trunk Sewer	Koshu	0.9 - 2.4	11.6
	Chino	0.7 - 1.2	6.8
	Chuoh	0.8 - 1.4	6.2
	Nishiyama	0.4 - 1.4	1.4
Sewage Pumping Station		2 Stations (Toyota & Shimosuwa)	
Sewage Treatement Plant		1 Plant (Toyota)	
Discharge Trunk	#1	1.5 - 4.5	4.7
	#2	0.8	4.3

The construction progress is shown in the following tables. Partial use of the system is expected by October, 1979.

TABLE 4. CONSTRUCTION PROGRESS OF LAKE SUWA BASIN SEWAGE SYSTEM (MILLION YEN)

Item	Total Construction	1971 - 1978	Remaining Construction	Progress %
Trunk sewer	12,789	6,664	6,125	52.1
Discharge trunk	4,952	448	4,504	9.0
Toyota sewage treatment plant	40,251	7,993	32,258	19.9
Shimosuwa sewage pumping station	1,023	713	310	69.7
Subcontracting, wages, etc.	2,985	2,212	773	74.1
Total	62,000	18,030	43,970	29.1

TABLE 5. CONSTRUCTION PROGRESS OF COMPLEMENTARY PUBLIC SEWAGE NETWORKS (1973-1978)

City	Constructed area (ha)	Distance (m)	Cost (million yen)	Total area (ha)	Progress (%)
Okaya	46.3	10,525	1,030	1,507.8	3.1
Suwa	108	19,685	1,743	1,423.4	6.2
Chino	37	6,634	467	714.7	5.2
Shimosuwa	26.8	4,772	466	515.6	5.2
Total	218.1	41,618	3,706	4,161.5	5.2

EFFLUENT CONTROL OVER INDUSTRY

Since completion of the sewage system is still several years away, local governments enacted ordinances setting their own environmental standards. These drainage standards required designated business firms in the basin area to meet the nation's environmental standard for lakes and marshes. Such standards are applicable to factories and offices whose average drainage per day exceeds 30 m^3. Local health centers are assigned to conduct on-site inspections to insure compliance.

DREDGING PROJECTS

Lake Suwa is a shallow lake with an average depth less than 4.0 m. Deposited on the bottom is a thick sediment layer resulting from the inflow of polluted water into the lake over a long period of time, and organic sludge from dead and decomposed aquatic plants and plankton. The adverse effects of these bottom sediments on the water quality is quite evident, particularly when sedimentary mud churned up by high waves creates water masses with a low oxygen content. This "Susumizu phenomenon" is doing extensive damage to the lake fishery. Dredging is therefore another fundamental step, along with the sewage system, towards regaining a clean lake. Table 6 presents environmental standards for lakes and marshes.

TABLE 6. ENVIRONMENTAL STANDARDS FOR LAKES AND MARSHES (Natural lakes and marshes and artificial lakes of at least 10 million m^3 capacity)

Quality		Standard					Water Body
		pH	COD	SS	DO	Coliform Count	
AA	City water 1st grade Fishery 1st grade Natural environment and Classes A, B, C	6.5 or above 8.5 or below	1 ppm or below	1 ppm or below	7.5 ppm or above	50 MPN/100 ml or below	
A	City water 2nd, 3rd grades Fishery 2nd grade Bathing and Classes B, C	6.5 or above 8.5 or below	3 ppm or below	5 ppm or below	7.5 ppm or above	1,000 MPN/100 ml or	Lake Suwa* Lake Shirakaba* Lake Tateshina*
B	Fishery 3rd grade Industrial water 1st grade Agricultural water and Class C	6.5 or above 8.5 or below	5 ppm or below	15 ppm or below	5 ppm or above	-	
C	Industrial water 2nd grade Environmental protection	6.0 or above 8.5 or below	8 ppm or below	No suspension of dust	2 ppm or above	-	

* entire lake

Basic Dredging Concepts

Basic concepts to support the dredging efforts are based on the following results from the studies made by the Lake Suwa Pollution Control Measures Investigation Committee:

1) Easily decomposed organic matter is deposited on shallow areas near shore.

2) Higher aquatic plants do not proliferate at water depths of over 2.5 m.

3) Dissolved oxygen (DO) at a depth of 2.5 m is higher than at lesser depths and almost equivalent to the DO at greater depths.

Thus, dredging at water depths of less than 2.5 m will be most effective.

Based on this conclusion a plan was drafted to build a new shoreline while retaining a substantial lake area of 13.3 km^2 for flood control (Figure 9). The old lake area was 14.06 km^2. The dredged sludge would be dumped in the area between the new and old shorelines and a greenbelt of streets and parks would be created by covering it with clean soil.

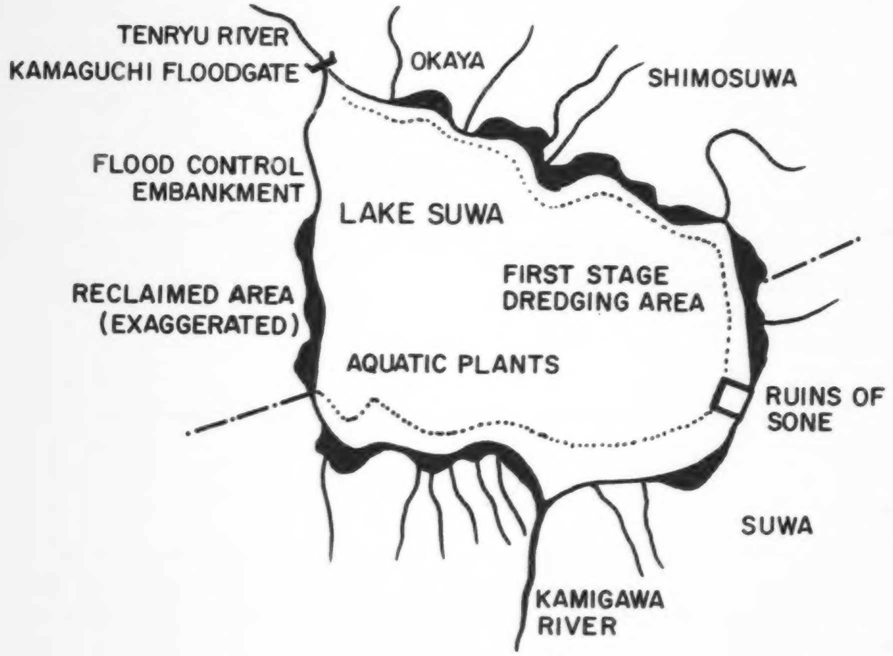

TENRYU RIVER
KAMAGUCHI FLOODGATE
OKAYA
SHIMOSUWA
FLOOD CONTROL EMBANKMENT
LAKE SUWA
FIRST STAGE DREDGING AREA
RECLAIMED AREA (EXAGGERATED)
AQUATIC PLANTS
RUINS OF SONE
SUWA
KAMIGAWA RIVER

Figure 9. Dredging and diking plan for Lake Suwa.

Dredging Plan

The dredging at Lake Suwa was initiated in 1969 as one of the "river environment protection projects" subsidized by the Japanese National Treasury (50% subsidy). The project is currently about 70 percent completed (Table 7).

TABLE 7. FIRST STAGE OF DREDGING PLAN

Item	Total Plan	1969 - 1977	Completion (%)
Project cost (million yen)	2,466	1,466	63.7
Dredged sludge (m^3)	1,656,000	1,211,860	73.2

The planned dredging area is about 2.5 km^2 or about 18 percent of the planned lake area of 13.3 km^2. About 78 percent of the dredged sludge was used as fill material between the new and old shorelines and 22 percent as top dressing to cover fields. This activity has been implemented with care to avoid secondary pollution.

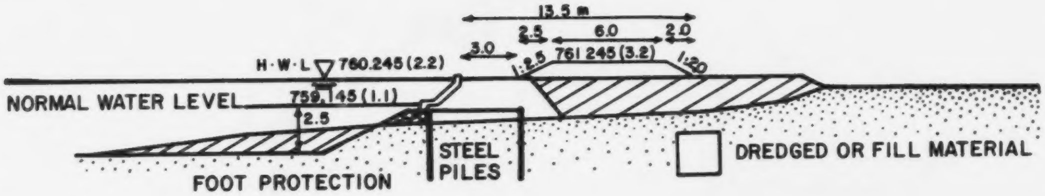

13.5 m
2.5
6.0
2.0
3.0
1:2.5
761.245 (3.2)
1:20
H·W·L 760.245 (2.2)
NORMAL WATER LEVEL
759.145 (1.1)
2.5
STEEL PILES
FOOT PROTECTION
DREDGED OR FILL MATERIAL

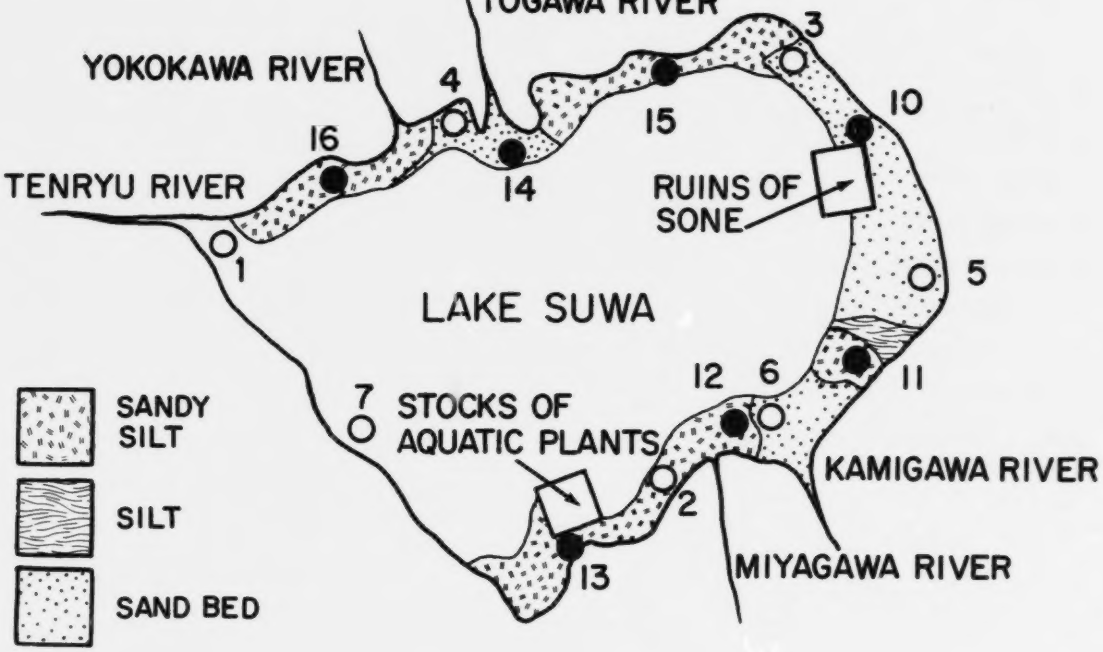
Diagram of dredge and backfill of new shorelines.

Dredged Sludge

The soil at the dredging area, typified by such characteristics as granular variation, hardness and specific gravity, affects the dredging ability. The results of soil examinations are shown in Table 8; test sites are identified in Figure 10.

TABLE 8. SOIL SAMPLE CHARACTERISTICS

	Sample No.						
Item	10	11	12	13	14	15	16
True specific gravity	2.517	2.615	2.693	2.419	2.828	2.627	2.629
Liquid limit	NP	NP	NP	-	NP	NP	NP
Plastic limit	NP	NP	NP	-	NP	NP	NP
Gravel content %	0	0	0	0	0	0	0
Sand content %	5.2	27	15.7	22	93	12	15
Silt content %	80.3	59.5	77.5	24	4	76	75
Clay content %	14.5	13.5	6.8	54	3	12	10
Max. grain size mm	0.84	0.84	0.42	0.42	-	-	-
Uniformity coefficient Uc	10	17	4	-	2.9	8.8	7.3
Curvature coefficient U'c	-	-	-	-	1.2	1.6	1.5
Soil type	silt	Psammitic silt	Psammitic silt	clay	sand	Psammitic silt	Psammitic silt
pH	7.2	5.8	6.4	-	-	-	-
Ignition loss %	5.1	11.5	9.1	-	-	-	-
Remarks	dark brown, offensive odor	dark brown, offensive odor	dark brown	dark brown, offensive odor	dark brown	dark brown, offensive odor	dark brown, offensive odor


TOGAWA RIVER
YOKOKAWA RIVER
TENRYU RIVER
3
4
10
15
16
14
1
RUINS OF SONE
5
LAKE SUWA
11
12
6
7
STOCKS OF AQUATIC PLANTS
KAMIGAWA RIVER
2
13
MIYAGAWA RIVER
SANDY SILT
SILT
SAND BED

● soil sample (see Table 8)

o core sample

Figure 10. Test sites.

Dredging Methods

For this project a pump dredge was selected from among the various types of available dredges. This was done for the following reasons:

1) Lake Suwa is a shallow inland lake.

2) There are many fishing facilities and ferry lines operating within the lake.

3) Pump dredging can be performed with a high mud content of the dredged material and without dispersing the sedimentary mud in the water.

Pipeline transportation of dredged material avoids emission of offensive odors. The pipelines were laid along the lake bottom to avoid obstructing vessels. Dredging equipment and pipeline specifications are shown in Tables 9 and 10 and Figure 11.

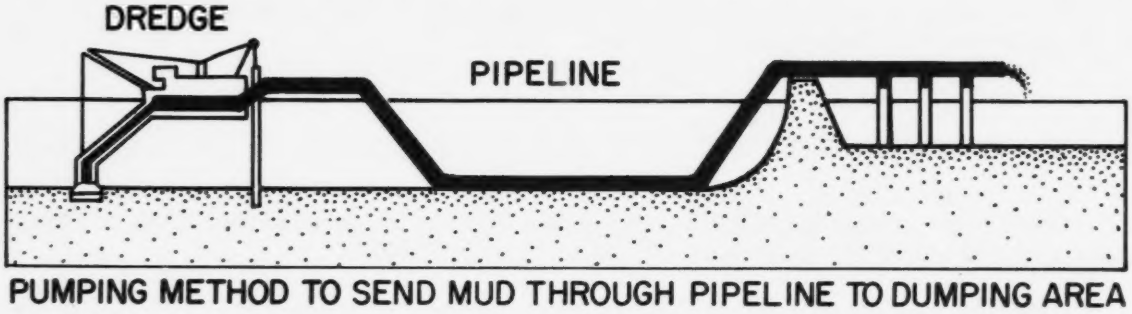


Figure 11. Pipeline system.

TABLE 9. DREDGING EQUIPMENT

Specifications	Main dredge (one)	Auxiliary dredge (one)	Relay boat (one)
Total weight (tons)	96	125	65
Type	Portable, on site fabrication	Portable, on site fabrication	Portable, on site fabrication
Length (m)	18.0	19.0	15.0
Width (m)	6.5	7.0	6.0
Draft (m)	1.0	1.0	0.8
Main engine	Electric motor (350 PS)	Diesel engine (420 PS)	Diesel engine (420 PS)
Dredging depth (m)	4.0	4.0	-
Max. transport distance (m)	900	900	600
Pumped water (m^3/h)	70	70	70
Total pump head (m)	45	45	45

Note: Mean mud content up to 50%
Sludge transport pressure 5 kg/cm^2

TABLE 10. PIPELINE SPECIFICATIONS

Pipeline pressure during sludge transportation: 50 kg/cm^2.
Flow rate in pipe during sludge transportation: 3.025 m/sec.
Diameter of sludge pipe: 310 mm
Length of sludge pipes in water (synthetic rubber pipes): 12 m/pipe.
Length of sludge pipes on land (cast iron pipes): 6 m/pipe.

Based on dredging results, (Figure 12), it is recommended that a relay boat be used where the transport distance is over 1,300 m.

Dredged sludge contains polluted water; when it is used as fill it can be a secondary source of pollution if allowed to return to the lake. To prevent this, the sludge is induced to settle rapidly at the reclamation area. The area between the new and old shorelines is divided by dikes into small ponds (average length: 50 m, width: 30 m, depth: 1.8 m). The sludge is pumped into these ponds and the water is removed. From the viewpoint of sedimentation efficiency, the best sludge transportation rate was 0.3 m/min or less. From the standpoint of an efficient dredging project, this rate was too slow and could not be tolerated. The relationship between grain size, settling time and depth is shown in Table 11.

TABLE 11. APPROXIMATE SETTLING TIME (HOURS)

Grain size / Settling depth	1.0 m	1.8 m
1.0 μ	3	3.6
0.7 μ	4	4.8

The local prefecture's SS criteria calls for 70 ppm. The national standard is 150 ppm (Water Pollution Prevention Act). Natural settling without coagulant agents produced 100 ppm SS discharges. Dredged material discharge to the settling ponds could not reasonably be reduced 0.3 m/min in order to meet the return flow SS criterion, therefore it was necessary to use a coagulant. The SS value was reduced to 50 ppm when 150 to 180 ppm of polyaluminum chloride coagulant was used.

Remaining Problems

About 10 years have passed since the first stage of the Lake Suwa Dredging Plan began in 1969. The project will be about 80 percent completed by the end of 1979. Even with the completion of the first stage in a couple of years, no noticeable improvement of the lake's water quality is expected, partly because the sewage system around the lake has not been completed.

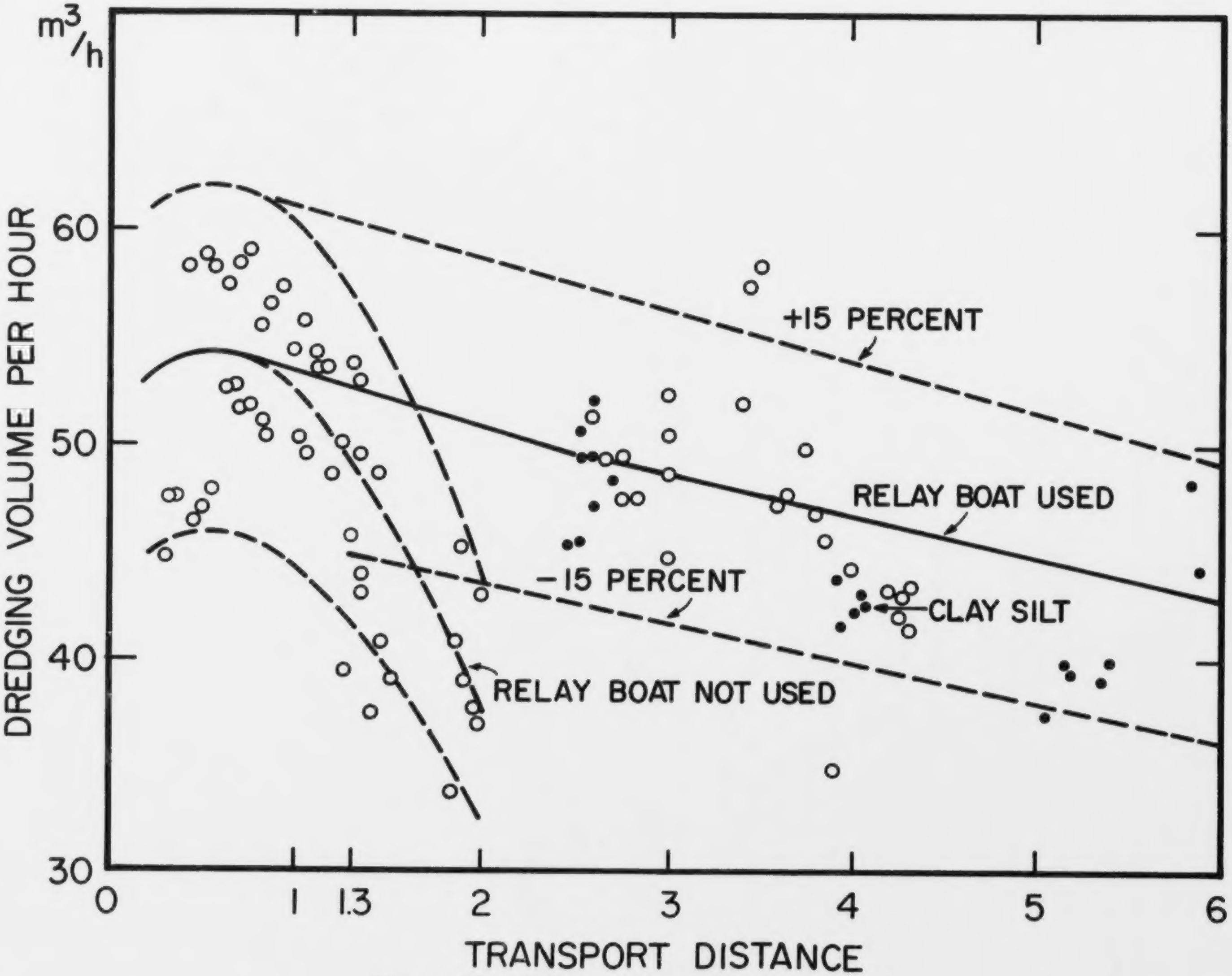


Figure 12. Dredging results (1973-1975).

Every summer the Aoko phytoplankton bloom that symbolizes the pollution of Lake Suwa turns the entire surface of the lake green. The Susumizu phenomenon, triggered by bottom sludge, still occasionally occurs. There is, however, a promising aspect to the project which requires further investigation. Up until 1963, the annual harvest of prawns from the lake averaged 8 tons. By 1968 it was reduced to zero. But since 1970, the second year of dredging work, the prawn catch has gradually increased to a current level of 3 tons per year.

Even when dredging is completed in all areas of the lake less than 2.5 m in depth, more than 80 percent of the deposited sludge will remain in the lake. It may still cause phytoplankton blooms and is therefore an obstacle to the rehabilitation of the lake.

Construction of the basin-wide sewage system (started in 1971) has been making steady progress and partial use of the system is expected in October of 1979. Its effect on the lake water will be known in a few years.

Consideration is also being given to expanding dredging to areas deeper than 2.5 m (over 80% of the total lake area). Before a final decision can be made it will be necessary to determine what the new sewage system can do. In addition, a study is currently being conducted on the flux of organic matter and nutrients in the lake.

What remains is to accurately measure the influence of bottom sludge on water pollution. A small portion of this work has already been finished and is reported in the following section.

The biggest problem to be solved in the future is how and where to dispose of newly dredged sludge, since the fill area between the new and old shorelines has been almost completely filled.

OUTLINE OF BOTTOM MATERIAL ANALYSIS

In 1976 and 1977 a study was made of the bottom sediments of Lake Suwa (Figure 13). It consisted of sediment analysis, elutriate testing, examination of mud suspension in the water column, and determination of sediment thickness. The idea was to determine the effects of the sediments on the water and the minimum dredging depth required to clean up the lake water.

LIST OF ANALYSES

a. Sediment analysis

Sample: 4 depths per station: 0, -30, -60, -90 cm

Total samples: 52

Analysis: Water content as percent dry weight, simple weight per volume, ignition loss, total carbon, total nitrogen, sulfide, COD.

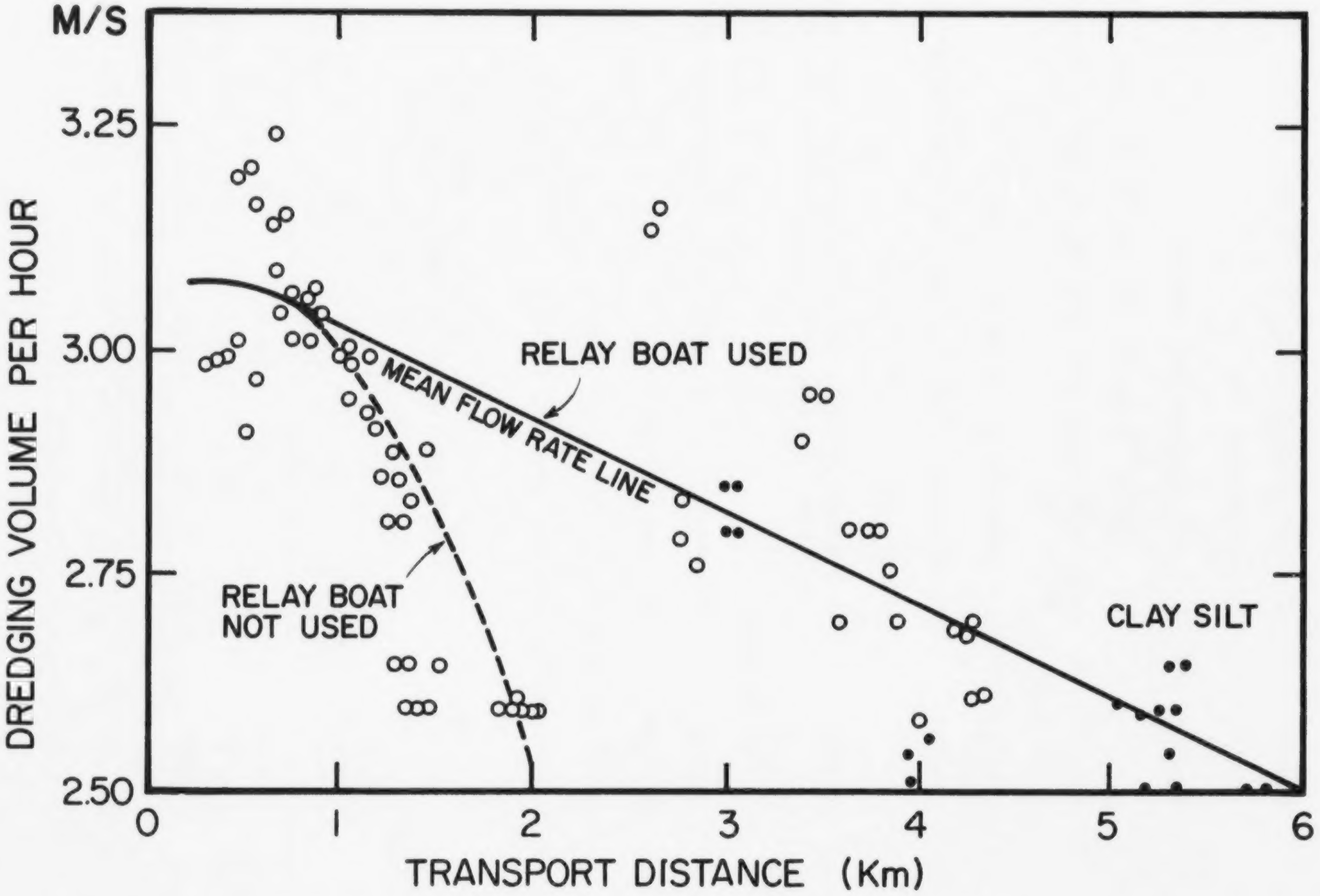


Figure 13. Dredging results (flow rate m/s) (1973-1975).

b. Elutriate test

Sample: 2 depths per station: 0, -60 cm

Total samples: 10

Analysis period: 10 analyses per sample:

Day 1, 3, 6, 10, 15, 20, 30, 45, and 60

Analysis: COD, TOC, total nitrogen, NH_4-N, total phoshporus, PO_4-P, DO

c. Analysis of mud suspension in the water column

Sample: 2 depths per station (upper and lower water layers)

Total samples: 16

Analysis: SS, water content, specific gravity, total phosphorus, total nitrogen, COD, NO_2-N, NO_3-N, NH_4-N, PO_4-P.

d. Sludge thickness

Device: Model RS-72 (Rasa Electronic Industries, Ltd.)

Frequency: 400 kHz, 30 kHz

Total distance traversed: 27.0 km

Results of the investigation are detailed in Tables 12 through 15.

TABLE 12. AVERAGE CONTENTS OF BOTTOM SEDIMENTS AT DIFFERENT DEPTHS IN THE UNDREDGED AREA

Depth	Water content (%)	Weight (g/cm^3)	Ignition loss (%)	TOC (mg/g)	T-N (mg/g)	Sulfide (mg/g)	COD (mg/g)
- 0 cm	473	1.14	16.3	42.2	4.49	1.10	30.1
-30 cm	349	1.17	16.4	40.8	4.00	1.40	31.1
-60 cm	282	1.27	14.7	35.5	3.41	0.91	18.5
-90 cm	180	1.37	13.5	26.3	2.73	0.53	15.8

TABLE 13. EFFECTS OF BOTTOM SEDIMENTS AT 0 cm AND -60 cm ON WATER QUALITY AS DETERMINED BY ELUTRIATE ANALYSIS AFTER 60 DAYS WHEN THE SEDIMENTARY MUD IS CONSIDERED MOST STABLE

Sample		COD (mg/l)	TOC (mg/l)	T-N (mg/l)	NH_4-N (mg/l)	T-P (mg/l)	PO_4-P (mg/l)	DO (mg/l)
0 cm sample	1	- 1.0	- 2.0	- 0.33	-	0.10	0.14	- 3.2
	2	- 0.4	-	0.30	-	0.06	0.15	- 2.9
	3	0.2	-	0.34	-	0.06	0.14	- 3.1
	4	0.2	1.0	- 0.77	-	0.01	0.13	- 4.0
	5	1.1	1.0	- 0.37	-	0.07	0.16	- 3.1
	Average	0.0	0.0	- 0.17	-	0.06	0.14	- 3.3
-60 cm sample	1	0.9	-	- 0.85	- 0.05	0.07	0.16	- 1.5
	2	0.2	2.0	- 1.30	-	0.08	0.16	- 1.3
	3	0.1	-	- 1.70	- 0.28	0.07	0.16	- 1.0
	4	1.3	1.0	- 1.14	- 1.26	0.10	0.16	- 1.0
	5	1.5	1.0	- 2.42	- 2.36	0.10	0.16	- 1.1
	Average	0.8	0.8	- 1.48	- 0.79	0.08	0.16	- 1.2
(-60 cm s) - (0 cm s)		0.8	0.8	- 1.31	- 0.79	0.02	0.02	2.1

TABLE 14. EFFECTS OF SUSPENDED MUD ON THE WATER COLUMN AS CALCULATED FOR THE AVERAGE WATER DEPTH OF 4 m

Above bottom	SS (mg/l·day)	Specific gravity	Inorganic				
			T-N (mg/l·day)	N (mg/l·day)	T-P (mg/l·day)	PO_4-P (mg/l·day)	COD (mg/l·day)
5.0 m	46.5	1.05	0.436	0.014	0.117	0.016	3.005
4.0	66.0	1.03	1.010	0.127	0.165	0.023	3.689
3.0	79.3	1.05	0.337	0.020	0.102	0.023	1.950
2.0	620	1.08	7.228	0.116	3.447	0.133	37.030
1.0	1,046	1.09	6.356	0.333	2.656	0.357	37.323
Average	372	1.06	3.073	0.122	1.297	0.110	16.599
per m^2 surface	kg/m^2·day 1.49	1.06	g/m^2·day 1.23	g/m^2·day 0.49	g/m^2·day 5.19	g/m^2·day 0.44	g/m^2·day 66.4

TABLE 15. EFFECTIVENESS OF DREDGING BASED ON THE RESULTS OF THE CURRENT INVESTIGATION AND EXISTING DATA

Data	Item	Effective depth	Effect
Current Investigation	DO	-60 cm	DO consumption reduced by 2 ppm
	Sulfide	-60 to -90 cm	Maximum at -30 cm. Almost same value at bottom surface and -60 cm. Reduced drastically at -60 to -90 cm, the value being below 1.0 mg/g at any point
	Ignition loss	-90 cm	No drastic reduction seen. 15% or below at -90 cm at most sites (11 out of 13 stations).
	COD	-30 to -60 cm	20 mg/g or below at -60 cm at all points.
	Suspended Mud	-30 cm	Returns to bottom material before "Susumizu phenomenon" occurs.
	Sludge Thickness	-20 to -60 cm	Eliminate sludge layer
Existing data	Methane Production	-20 cm	Reduced bottom surface production to 1/10.

RESULTS OF ANALYSIS

1. In the elutriate test, the materials at 0 cm and 60 cm in depth were sampled at 5 points. About 150 g of evenly mixed mud was put in 1-l dark bottles with 700 ml of filtered lake water (5A filter paper). Dark bottles were also filled with filtered lake water only as controls. Both types of bottles were sealed and retained for testing. Water analyses of both bottle types were made on each specified day.

Lake water sampled for the test (on August 3rd and 4th, 1977) contained little NH_4-N. Most of the total nitrogen is considered to exist in an organic state. This may be due to nitrogen uptake by Aoko (Microcystis) in the upper layers.

About 80-100% of the phosphorus in the water existed in PO_4-P. Microcystis growing rapidly in the summer quickly takes up PO_4-P as well as NH_4-N. The fact that PO_4-P existed with little NH_4-N and NO_3-N in the lake water suggests that N is the limiting factor in the propagation of phytoplankton in the upper water layer of the lake, at least at the time of sampling (Table 16).

TABLE 16. ANALYSIS OF RESULTS OF CONTROL BOTTLES OF FILTERED LAKE WATER (5A filter paper)

C.O.D. (mg/l)	T.O.C. (mg/l)	T - N (mg/l)	NH_4-N (mg/l)	T - P (mg/l)	PO_4-P (mg/l)	DO (mg/l)
3.0	8	1.99	0.01 or above	0.19	0.19	7.2

2. The methodology used in the mud tests was proposed by Dr. Koyama of Nagoya University and others. Sampling devices were made and used for one month. Two of the devices sustained damage from ferryboats. At three of six stations, a large quantity (15-8 cm) of particles with specific gravity greater than 1.1 were suspended 1 to 2 m above the bottom. At the other three points the SS layer was much less (less than 3 cm).

The 1.0 to 2.0 m thick suspended solid layers contained 600 to 1000 ppm of SS. Once resuspended, this mud suspension does not settle easily. Once in suspension it can migrate to higher layers or to the surface.

The thickness of bottom sediments contributing to the dense overlying layers of suspended mud was calculated from the data to be less than 1 cm. In the lake it would be a few centimeters at most. Since Lake Suwa is very shallow with an average depth of about 4 m and a maximum depth of 6.8 m, wind-blown waves can easily reach the bottom and cause mud resuspension.

3. The results of sludge thickness measurements showed that the sludge in the dredged area was about 10 cm thick. That of the undredged area was 20 to 60 cm.

4. The bottom sediments of Lake Suwa display very small unit weight and grain size. Consequently, dredging to a depth of -90 cm may not effectively prevent mud resuspension. But the Susumizu phenomenon caused by suspended mud did not occur before 1966. The rate of sedimentation in Lake Suwa is 5 to 20 cm over 10 years. Considering that the sediment thickness which causes mud resuspension is only a few centimeters, we estimate about 30 cm of dredging will restore the bottom to 1965 conditions, when the Susumizu phenomenon did not occur.

CONCLUSION

More than a decade has passed since water quality control efforts began in Lake Suwa, one of the most eutrophic lakes in Japan. During this period, scientific investigations have been conducted and administrative measures taken. But pollution continues and algal blooms occur yearly.

However, the construction of the Lake Suwa Basin Sewage System has made steady progress since its beginning in 1971. Partial use of the system will begin in October, 1979. Progress is also being made in the river improvement projects around the lake and in dredging of the lake itself. But a method for total rehabilitation of the lake has not yet been found. Research is now being done to determine the mechanisms of organic pollution and the methods appropriate to predict the behavior of pollutants in the lake.

It will take a long time, but all the projects in Lake Suwa are making progress toward restoring the lake to cleaner, purer water quality.

REFERENCES

(1) Lake Suwa Purification Measures Investigation Committee, "Studies on Purification of Lake Suwa".

(2) Nagano Research Institute for Health and Pollution, Pollution Section, Nagano Prefecture Department of Living Environment, "Report on Investigation of Countermeasures against Eutrophication of Lake Suwa".

(3) The Civil Engineering Management Office for Lake Suwa and Surrounding Region, "Examination of Lake's Bottom Material as Part of River Environment Improvement Works".

(4) Nagano Prefecture, "Pollution White Paper".

DREDGING OF POLLUTED BOTTOM SEDIMENTS IN THE IBO RIVER

Sadao Kishimoto
Director of Himeji Work Office
Kinki Regional Construction Bureau
Ministry of Construction, Japan

INTRODUCTION

Located in the western part of Hyogo Prefecture, the Ibo River flows into the Seto Inland Sea at Himeji City - about 80 kilometers west of the International Port of Kobe. This region is located along the Sanyo Route, historically an important connection between the western part of mainland Japan and Kyushu Island, Shikoku Island and the Chugoku Region.

Various industries have developed in the basin, notably brewing, the production of Japanese noodles, and leather manufacturing. The leather manufacturing factories along the Hayashida River, a tributary of the lower main channel of the Ibo, have created a serious water pollution problem in both rivers. This has also caused a large volume of bottom sediments containing polluted substances to accumulate on the river beds. In the dry season, unavoidable impacts from this pollution are found in the form of foul odors and other negative influences on irrigation, fishing and the general environment. The volume of polluted bottom sediments is estimated at 8,000 m^3 at the confluence of the Ibo and its tributary, the Hayashida River. The total volume, including that in the river estuary, is estimated at several tens of thousands of cubic meters of sediment.

To deal with this problem, stricter regulation of wastewater discharge, the construction and improvement of sewage treatment facilities and dredging of bottom sediment are now being adopted. This paper describes the plan for dredging the bottom sediments.

THE OUTLINE OF THE BASIN

The Ibo River is shown in Figure 1. Its headwaters begin at Mt. Fujinashi (1,139 m) in the Chugoku mountain chain. As it flows to the Seto Inland Sea it is joined by major tributaries - the Hikihara, Kurusu, and Hayashida Rivers. The area of the Ibo basin is approximately 810 km^2, and the length of the main river channel is 71 km.

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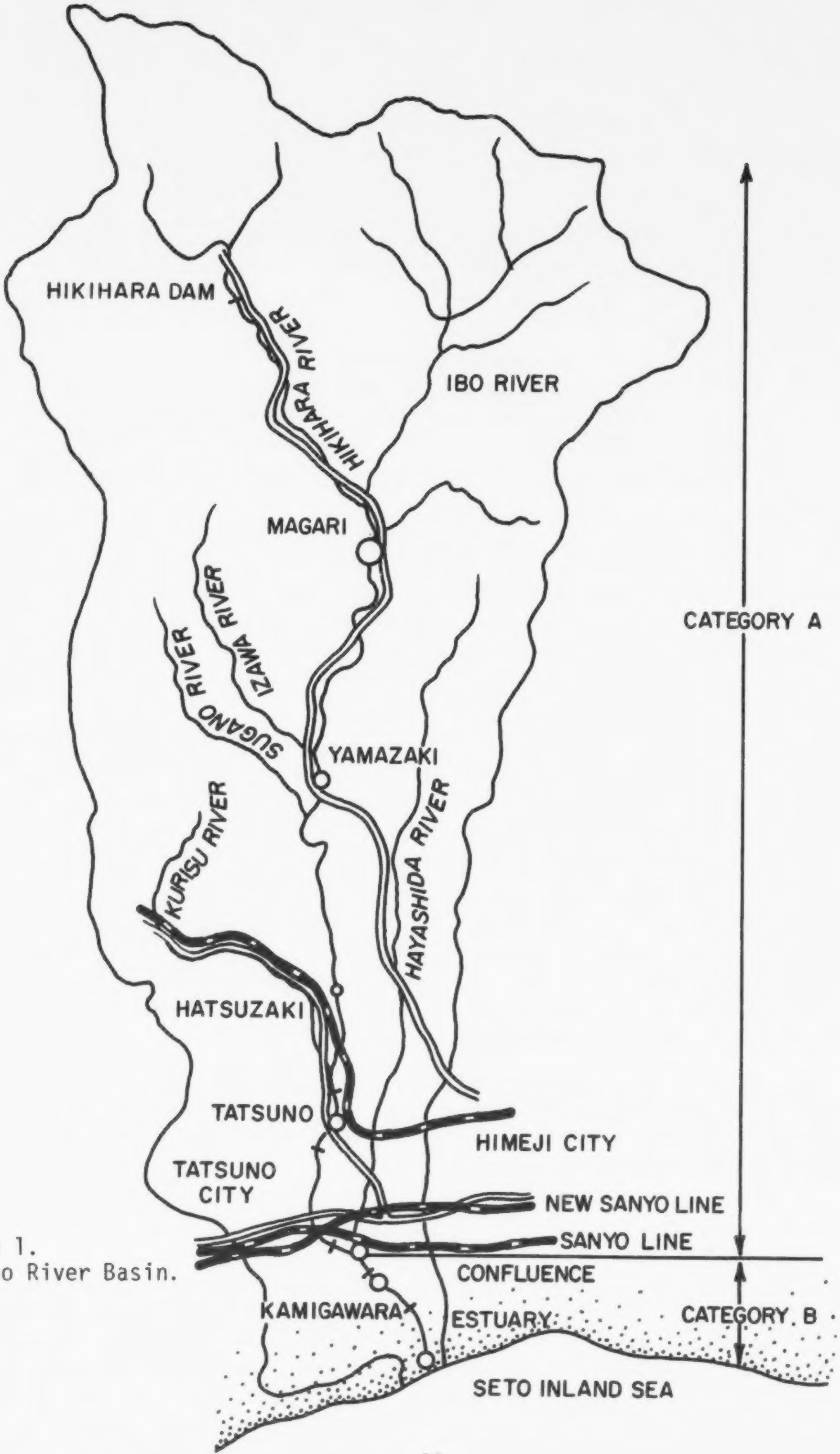

HIKIHARA DAM
HIKIHARA RIVER
IBO RIVER
MAGARI
IZAWA RIVER
SUGANO RIVER
YAMAZAKI
KURISU RIVER
HAYASHIDA RIVER
HATSUZAKI
TATSUNO
HIMEJI CITY
TATSUNO CITY
NEW SANYO LINE
SANYO LINE
CONFLUENCE
KAMIGAWARA
ESTUARY
SETO INLAND SEA
CATEGORY A
CATEGORY B

Figure 1.
The Ibo River Basin.

The major part of the basin is situated in mountainous terrain - the flat area is only 170 km^2, about 20% of the total area. Hayashida Watershed occupies 97 km^2 of the basin.

The geology consists of basaltic andesite and quartz diorite in the mountains and alluvial strata in the valley. The total forest area of 640 km^2 is composed of 190 km^2 coniferous forest, 260 km^2 deciduous forest and 190 km^2 brush.

Annual average rainfall in the basin is about 2100 mm upstream, about 1600 mm in the middle stream and about 1400 mm downstream. The riverbed slope is 1/2000 to 1/100 in the Ibo River and 1/360 to 1/280 in the Hayashida.

Table 1 shows the 18 year average stream flow at Tatsuno in the main river. Table 2 shows the monthly mean flow at Tatsuno, Kamigawara (Ibo River) and Kamae (Hayashida River) for 1975 and 1976. As these tables indicate, heavy flow is expected in the rainy season, and zero flow in the dry season.

The watershed area of the Hayashida River is affected by the climate of the Seto Inland Sea. Its discharge per unit drainage area is roughly 70% of that of the main river.

The Ibo River drainage area contains two cities and eight towns, total population about 200,000. Downstream the Harima seaside industrial zone is located, an area of extensive chemical and heavy industries. From Tatsuno City to Himeji City brewing, noodle and leather industries are dominant. These industries are major sources of pollution, particularly the leather industry.

The present water rights in the Ibo River are 0.09 m^3/sec (0.2%) for city water use, 5.15 m^3/sec (15.9%) for industrial water supply, 16.98 m^3/sec (52.4%) for agricultural use and 10.20 m^3/sec (31.5%) for power. The maximum volume is 32.41 m^3/sec. The water withdrawn below the Hayashida River confluence is used for agricultural and industrial purposes, thus water quality is a major concern.

PRESENT CONDITION OF WATER QUALITY AND BOTTOM SEDIMENTS IN THE IBO RIVER

WATER QUALITY OF THE IBO RIVER

In May 1973, the law which set environmental standards was applied to the Ibo River, and the stream above the Hayashida confluence was classified as category A and the stream below as category B, (Figure 1, also Table 3 for A and B).

Since the biological oxygen demand (BOD), suspended sediments (SS) and dissolved oxygen (DO) values in the middle and upstream Ibo are respectively 1.0-1.5 mg/l, 13-19 mg/l and 9.9-10.4 mg/l, there are few pollution problems there. Therefore this discussion will concentrate on the severely polluted tributary, the Hayashida River and its confluence with the main river.

Three bottom sediment monitoring stations, shown in Figure 2 were established: Tatsuno station in the Ibo River, Kamae station in the Hayashida

TABLE 1. STREAM FLOW GIVEN AS MEAN VALUE OVER 18 YEARS AS MEASURED AT TATSUNO STATION (m^3/sec).

Flow	Max.	95th day	185th day (mean flow)	275th day (mean flow)	355th day	Min. flow
Mean value	1,152	27.6	16.9	11.1	4.9	2.5

TABLE 2. MONTHLY MEAN FLOW (m^3/sec).

Moni-toring Station	Year	Month												Mean Value
		Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
	1975	14.83	27.34	37.66	58.25	41.35	39.60	54.38	28.18	12.34	36.03	36.38	15.84	33.53
	1976	9.25	31.94	28.31	55.61	42.67	38.55	13.54	29.16	175.31	20.65	19.17	15.02	39.93
	1975	11.14	18.67	23.73	32.50	22.27	22.35	28.58	15.59	8.24	24.38	21.64	11.01	20.01
	1976	6.68	18.01	18.08	24.44	20.59	16.33	7.11	14.21	66.60	10.29	5.82	2.30	17.54
	1975	2.28	2.49	2.49	5.60	5.29	6.24	5.60	2.49	1.40	4.99	3.38	0.85	3.59
	1976	0.66	0.88	1.41	3.28	4.25	4.25	1.56	2.84	25.56	3.84	2.59	3.57	4.56

TABLE 3. ENVIRONMENTAL STANDARDS FOR RIVERS

		Standard Values[1]				
	Purpose of Utilization	pH	Biological Oxygen Demand (BOD)	Suspended Solids (SS)	Dissolved Oxygen (DO)	Number of Coliform Groups
AA	Water supply, class 1, conservation of natural environment and uses	6.5 - 8.5	1 ppm or less	25 ppm or less	7.5 ppm or more	50 MPN/100 ml or less
A	Water supply, class 2; fishery, class 1; bathing and uses listed in B - E	6.5 - 8.5	2 ppm or less	25 ppm or less	7.5 ppm or more	1,000 MPN/100 ml or less
B	Water supply, class 3; fishery, class 2, and uses listed in C - E	6.5 - 8.5	3 ppm or less	25 ppm or less	5 ppm or more	5,000 MPN/100 ml or less
C	Fishery, class 3; industrial water class 1, and uses listed in D - E	6.5 - 8.5	5 ppm or less	50 ppm or less	5 ppm or more	
D	Industrial water, class 2; agricultural water[2], and uses listed in E	6.0 - 8.5	8 ppm or less	100 ppm or less	2 ppm or more	
E	Industrial water, class 3; conservation of environment	6.0 - 8.5	10 ppm or less	Floating matter such as garbage should not be observed	2 ppm or more	

[1] The standard value is based on the daily average value. (The same applies to the standard values for lakes and coastal waters.)

[2] At the supply inlet for agricultural water, pH shall be between 6.0 and 7.5 and dissolved oxygen shall not be less than 5 ppm. (The same applies to the standard values for lakes.)

TABLE 3. (continued)

Notes:

1. Conservation of natural environment: Conservation of scenic spots and other natural resources.

2. Water supply, class 1: Water treated by simple cleaning operations such as filtration.
   Water supply, class 2: Water treated by normal cleaning operations such as sedimentation and filtration.
   Water supply, class 3: Water treated through a highly sophisticated cleaning operation including pretreatment.

3. Fishery, class 1: For aquatic life such as trout and bull trout inhabiting oligotrophic water, and those of fishery class 2 and class 3.
   Fishery, class 2: For aquatic life, such as fish of the salmon family and sweetfish inhabiting oligotrophic water and those of fishery class 3.
   Fishery, class 3: For aquatic life such as carp and silver carp inhabiting β-mesotrophic water.

4. Industrial water, class 1: Water given normal cleaning treatment such as sedimentation.
   Industrial water, class 2: Water given sophisticated treatment by chemicals.
   Industrial water, class 3: Water given special cleaning treatment.

5. Conservation of environment: Up to the point at which no unpleasantness is caused to people in their daily lives (including a walk by the riverside).

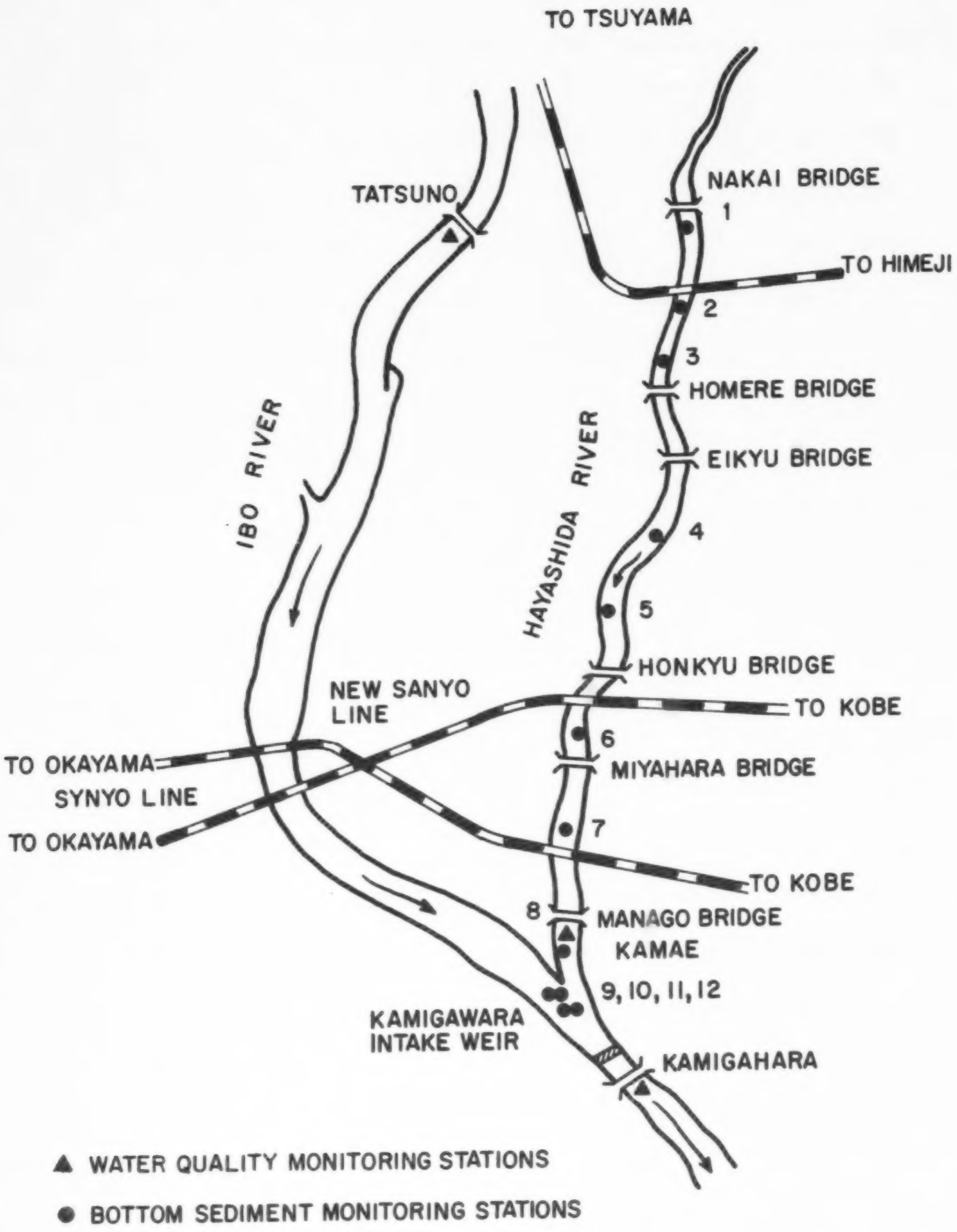

TO TSUYAMA
TATSUNO
NAKAI BRIDGE
1
TO HIMEJI
2
3
HOMERE BRIDGE
EIKYU BRIDGE
IBO RIVER
HAYASHIDA RIVER
4
5
HONKYU BRIDGE
NEW SANYO LINE
TO KOBE
6
MIYAHARA BRIDGE
TO OKAYAMA
SYNYO LINE
TO OKAYAMA
7
TO KOBE
8
MANAGO BRIDGE
KAMAE
9, 10, 11, 12
KAMIGAWARA INTAKE WEIR
KAMIGAHARA
▲ WATER QUALITY MONITORING STATIONS
● BOTTOM SEDIMENT MONITORING STATIONS

Figure 2. Major monitoring stations.

TABLE 4. THE CHANGE IN WATER QUALITY WITH TIME AS ANNUAL MEAN

	Item	Unit	Year 1971	1972	1973	1974	1975	1976	1977
Ibo River (Tatsuno)	pH		7.5	7.4	7.4	7.4	7.3	7.4	7.4
	DO	mg/l	9.8	10.4	10.8	10.0	10.1	9.9	10.4
	COD	mg/l	3.2	3.0	1.5	1.8	0.9	1.6	1.4
	BOD	mg/l	3.1	2.0	1.8	1.6	1.5	1.5	1.8
	SS	mg/l	10	7	12	17	12	19	7
	Cl^-	mg/l	12.7	16.3	8.8	6.8	6.4	5.8	6.2
	NH_4-N	mg/l	0.07	0.12	0.15	0.11	0.09	0.06	0.11
	NO_2-N	mg/l	0.020	0.038	0.029	0.008	0.008	0.009	0.012
	NO_3-N	mg/l	0.20	0.26	0.43	0.53	0.79	0.58	0.68
	NH_4-N+ NO_2-N+ NO_3-N	mg/l	0.290	0.418	0.609	0.648	0.888	0.649	0.802

	Item	Unit	Year 1971	1972	1973	1974	1975	1976	1977
Hayashida River (Kamae)	pH		10.3	10.5	10.8	10.2	10.4	10.7	10.6
	DO	mg/l	0.9	2.1	1.3	1.8	1.0	0.3	0
	COD	mg/l	148	81.4	72.1	62.4	65.9	123	204
	BOD	mg/l	233	145	223	268	182	231	236
	SS	mg/l	172	99	156	142	252	248	372
	Cl^-	mg/l	472	342	434	377	399	458	710
	NH_4-N	mg/l	28.0	10.8	15.7	15.0	13.9	16.6	23.6
	NO_2-N	mg/l	0.048	0.101	0.012	0.057	0.093	0.028	0.013
	NO_3-N	mg/l	0.19	0.18	0.18	0.34	0.93	0.01	0.15
	NH_4-N+ NO_2-N NO_3-N	mg/l	28.238	11.081	15.892	15.397	14.923	16.638	23.763

TABLE 4. (continued)

	Item	Unit	Year 1971	1972	1973	1974	1975	1976	1977
	pH		7.8	8.5	9.2	8.2	8.3	8.6	8.4
	DO	mg/l	7.2	7.5	8.1	8.5	8.0	7.5	7.8
Ibo River (Kamigawara)	COD	mg/l	8.0	6.8	8.5	7.6	6.8	11.1	10.8
	BOD	mg/l	8.8	13.2	14.8	14.4	13.5	19.6	18.8
	SS	mg/l	15	12	12	19	13.5	19.6	18.8
	Cl^-	mg/l	34.6	31.1	42.1	42.9	71.5	53.0	41.6
	NH_4-N	mg/l	0.56	0.95	2.13	1.34	2.12	1.95	1.67
	NO_2-N	mg/l	0.232	0.097	0.072	0.029	0.048	0.025	0.036
	NO_3-N	mg/l	0.22	0.24	0.26	0.56	0.62	0.10	0.59
	NH_4-N+ NO_5-N+ $NO_{2/3}$-N	mg/l	1.012	1.257	2.462	1.929	2.788	2.075	2.296

River and Kamigawara station downstream from the confluence. Average annual values for environmental parameters of these three stations are shown in Table 4 and Figure 3.

As shown, the water quality of the Hayashida River has badly deteriorated, which results in the deterioration of water quality in the downstream Ibo River. The pH at the Tatsuno station beyond the influence of the polluted Hayashida River is a stable 7.3-7.5. At the Kamae station, the pH value is 10.2-10.8. At the Kamigawara station downstream, the value fluctuates between 7.8-9.2.

The maximum average annual DO value at the Kamae station was 2.1 mg/l in 1972. Since then, there has been a large amount of oxygen consumed by organic pollutants. Except for a peak of 1.8 mg/l DO in 1974, it has constantly decreased to almost 0 mg/l in 1977. At the Tatsuno station, the DO value has stayed high at 9.8-10.8 mg/l. At Kamigawara it has declined to 2.0 mg/l lower than at Tatsuno.

The BOD value at Tatsuno was at its peak of 3.1 mg/l in 1971 and since then has been constantly decreasing. At Kamae, it is 145-268 mg/l, which corresponds to sewage influx. No clear trend can be observed due to large fluctuations of BOD. At Kamigawara, BOD reached a minimum of 8.8 mg/l in 1971. Since then it has increased rapidly to 20 mg/l in 1976 and 1977.

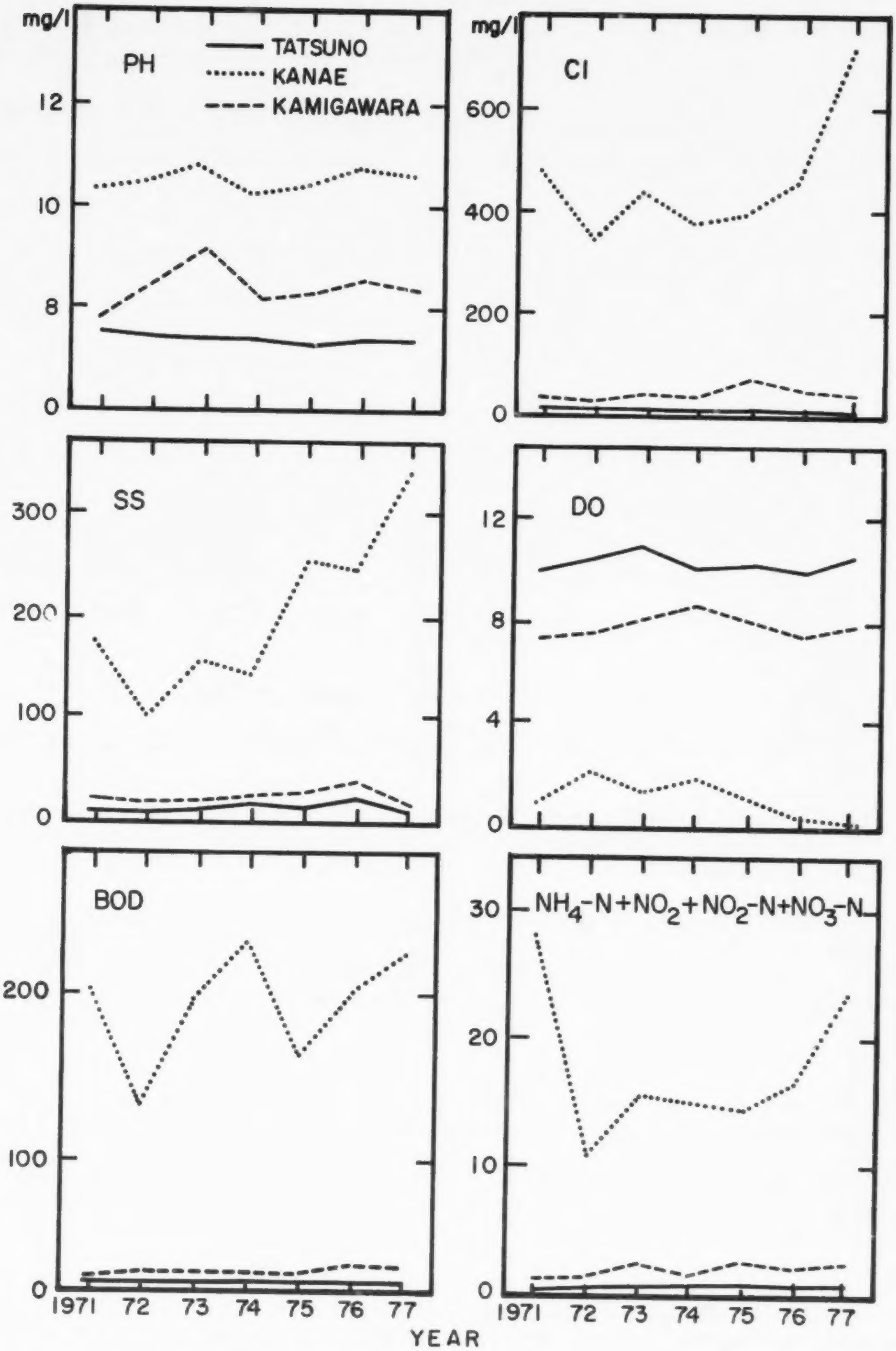


Figure 3. The change in water quality over time (annual mean value).

The SS value at Tatsuno, low at 7-19 mg/l, shows no tendency to change. At Kamae, SS levels ranged between 99-172 until 1974. Since then the level has increased rapidly to 372 mg/l in 1977. At Kamigawara, SS levels were a little higher (by 5-10 mg/l) than at Tatsuno, but compared to the other environmental factors little influence from the Hayashida River is observed. This may be caused by the effects of settling as the river flows downstream to the station.

The Cl^- value at Tatsuno is 5.8-16.3 mg/l. Its average value over the past seven years has been 9 mg/l, which is normal river water quality. The minimum value at Kamae was 342 mg/l in 1971, but since then has increased to almost twice that figure (710 mg/l in 1977). This was caused by wastewater discharge from the leather industry where salt was being washed from hides. The Cl^- is different from other factors since it is not physically and biologically decomposed. The Kamigawara station was directly affected by the Hayashida River and after 1975, the chloride at Kamigawara increased to about ten times that of Tatsuno.

Inorganic nitrogen values for NH_4-N and NO_2-N at Tatsuno are relatively small, but NO_3-N has increased rapidly from its minimum of 0.2 mg/l in 1971. NO_3-N is the final decomposition product for inorganic nitrogen. The ratio of NO_3-N to total inorganic nitrogen is largest at Tatsuno. In contrast, inorganic nitrogen at Kamae is 11-28 mg/l, of which NH_4-N is more than 90%. The NO_3-N at Kamigawara is almost the same as that of Tatsuno, but the NH_4-N increases by 1.5-2.0 mg/l due to the influence of the Hayashida River. From these facts, it may be concluded that biodegradation does not reach the nitrification stage during the 1/2 km it takes the river to flow from Kamae to Kamigawara.

Another investigation of water quality conducted on the Hayashida River showed that the total phosphorus (0.2-1.7 mg/l) is relatively small compared with BOD. This may be caused by domestic wastewater. But total chrome is also relatively high at 1-10 mg/l of discharged water. Thus, the water pollution in the Hayashida River is mainly caused by the leather industry.

BOTTOM SEDIMENTS IN THE IBO RIVER

Since there is no specific pollution problem in the middle and upstream of the Ibo River, the discussion will be confined mainly to the Hayashida River and the confluence with the main channel where the water is badly polluted by the leather industry.

In this investigation, pH, COD, volatile and fixed material, total nitrogen, total phosphorus, sulfide, total chromium, water content and grain size composition were measured as bottom sediment characteristics. PCB, total mercury, alkyl-mercury, cadmium, lead, cyanogen $(CN)_2$, arsenic, hexavalent chromium and organic phosphate were measured in elutriate tests.

The Hayashida River has a steep bed with a slope of about 1/300. The flow velocity is very high. Consequently, suspended solids or minute sludges may not settle, but organic sludges generated from polluted wastewater will accumulate at weirs and at the confluence where the velocity is reduced.

The "sludge" component of the bottom sediments has a low specific gravity of 1.78-1.90. The water content is high at 63.1-90.3%. $m = \frac{Ww}{wt}$ where:

m = water content, wet wt. basis
Ww = weight of water
Wt = total wt. of soil

Clay and silt with a diameter of less than 74 μ make up 75-95% of the sludge. The unit volume weight is small at 1.02 g/cm^3.

The sand-gravel component in the bottom sediments is 65-94% gravel. Specific gravity is in the range of 2.64-2.68.

Figure 4 shows the relationship between COD and volatile and fixed matter in the bottom sediments of the Hayashida River. The volatile and fixed suspended matter in the sludge differs from that in the sand-gravel component. In the former, it varies from 8.8% to 68.6%, but most fall into the range of 40-70%.

The COD value is also separated into the two components. It is in the range of 11 mg/g to 290 mg/g dry weight in the sludge but mostly in the range of 40-300 mg/g. These results show that the bottom sediments of the Hayashida River are composed of organic sludge with a large amount of volatile and fixed matter and COD.

Sulfide content is small (about 40 mg/kg, dry weight) upstream of the Hayashida River. It increases to 417 mg/kg in the lower stream and near the confluence it is 1,090 mg/kg. This shows that reduction is occurring during stream flow.

Depending upon the location, the total chromium content is 1,300 mg/kg to 21,000 mg/kg. This is caused by the leather industry wastewater which includes protein compounds and chromium resulting from the leather tanning process. High nitrogen content of 605-22,100 mg/kg dry weight and phosphorus content of 850-2,780 mg/kg also exists.

The results of the elutriate tests showed the leaching rate to be lower than the limit of quantative analysis for all the tested items, as shown in Table 5.

THE WATER POLLUTION MECHANISM AND CONTROL MEASURES CRITICAL IN THE IBO RIVER

WATER POLLUTION MECHANISM

As discussed in the previous section, water pollution in the Ibo is caused by wastewater from industry along the Hayashida River. Bottom sediments accumulate around weirs and at the confluence of the two rivers where flow velocity is reduced.

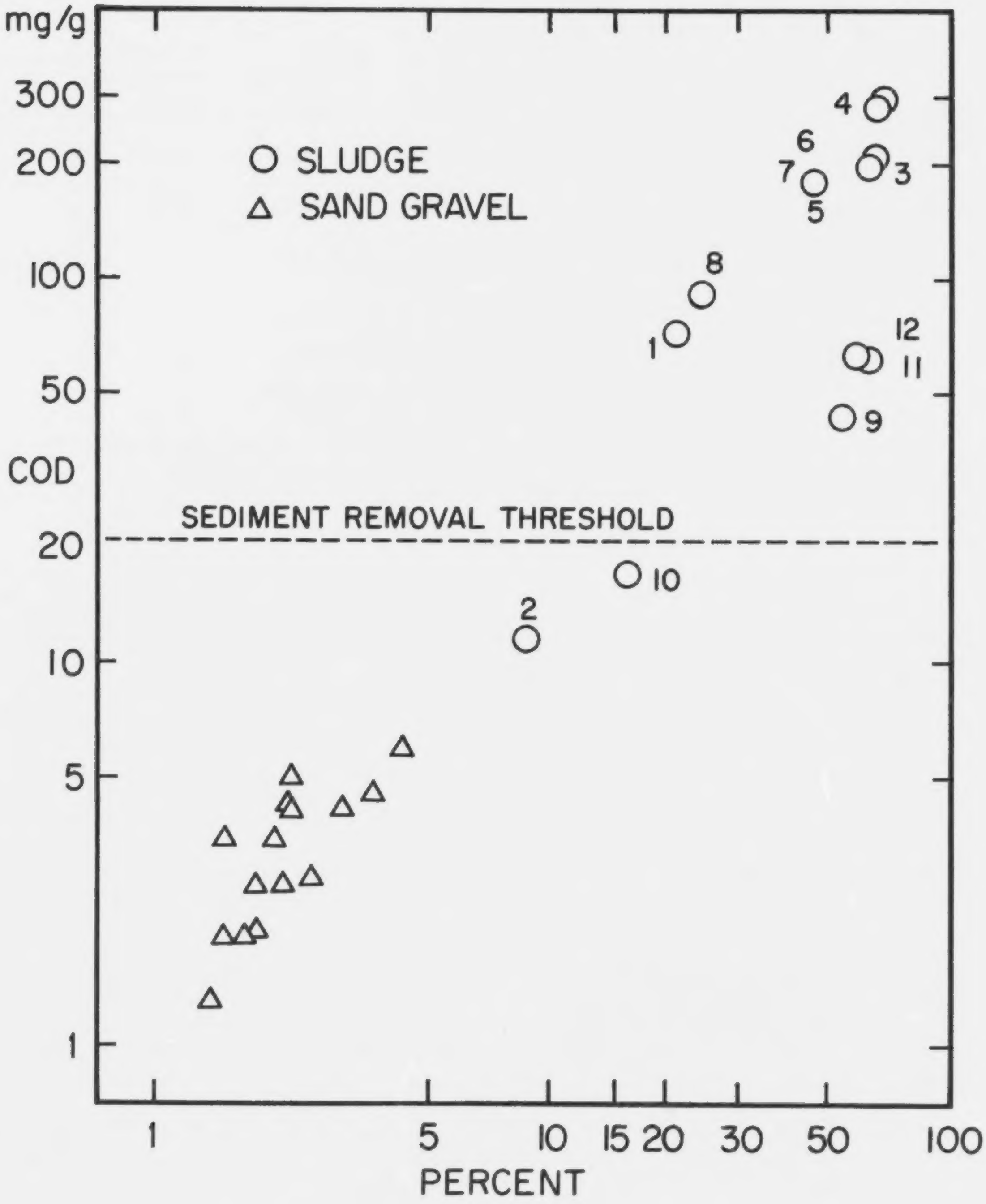


Figure 4. COD and volatile and fixed material in sediments.

TABLE 5. ELUTRIATE TEST OF SURFACE SEDIMENT

Item		Monitoring Point		Limit of Quantative Analysis	Minimum Acceptable Level
		Hayashida River	Confluence of the Hayashida and Ibo River		
T - Hg	mg/l	not detected	not detected	0.0005	0.005
R - Hg	mg/l	not detected	not detected	0.0002	N.D.
Cr^{6+}	mg/l	not detected	0.0005	0.005	0.5
Pb	mg/l	not detected	not detected	0.01	1
Cd	mg/l	not detected	not detected	0.005	0.1
As	mg/l	not detected	not detected	0.01	0.5
Org. Phosphate	mg/l	not detected	not detected	0.005	1
CN	mg/l	not detected	not detected	0.03	1
PCB	mg/l	not detected	not detected	0.005	0.003

Ref: The method for performing tests is given in notice No. 14 of the Environmental Agency.

The poor water quality of the Hayashida affects that of the lower main Ibo and polluted sludge moved by floods further aggrevates the water quality of the lower stream. The deterioration in the river water, together with polluted bottom sediments, has affected agriculture and industry. It causes eutrophication, DO reduction and some anoxic conditions which result in the death of aquatic organisms. This pollution mechanism is diagrammed in Figure 5.

A variety of approaches should be followed in cleaning up the river, including restriction on discharge of industrial wastewater, installation of sewage treatment facilities and industrial waste, introduction of clean water to the polluted river and dredging of bottom sediments to enhance the capacity of the river for self-renewal.

These steps have been or are now being taken. Wastewater discharge restrictions have been applied to all local industry and installations of pre-treatment facilities are now occurring.

In Tatsuno City, the construction of public sewage facilities is making steady progress. In the upper Hayashida, construction of a dam has been launched to control the stream flow.

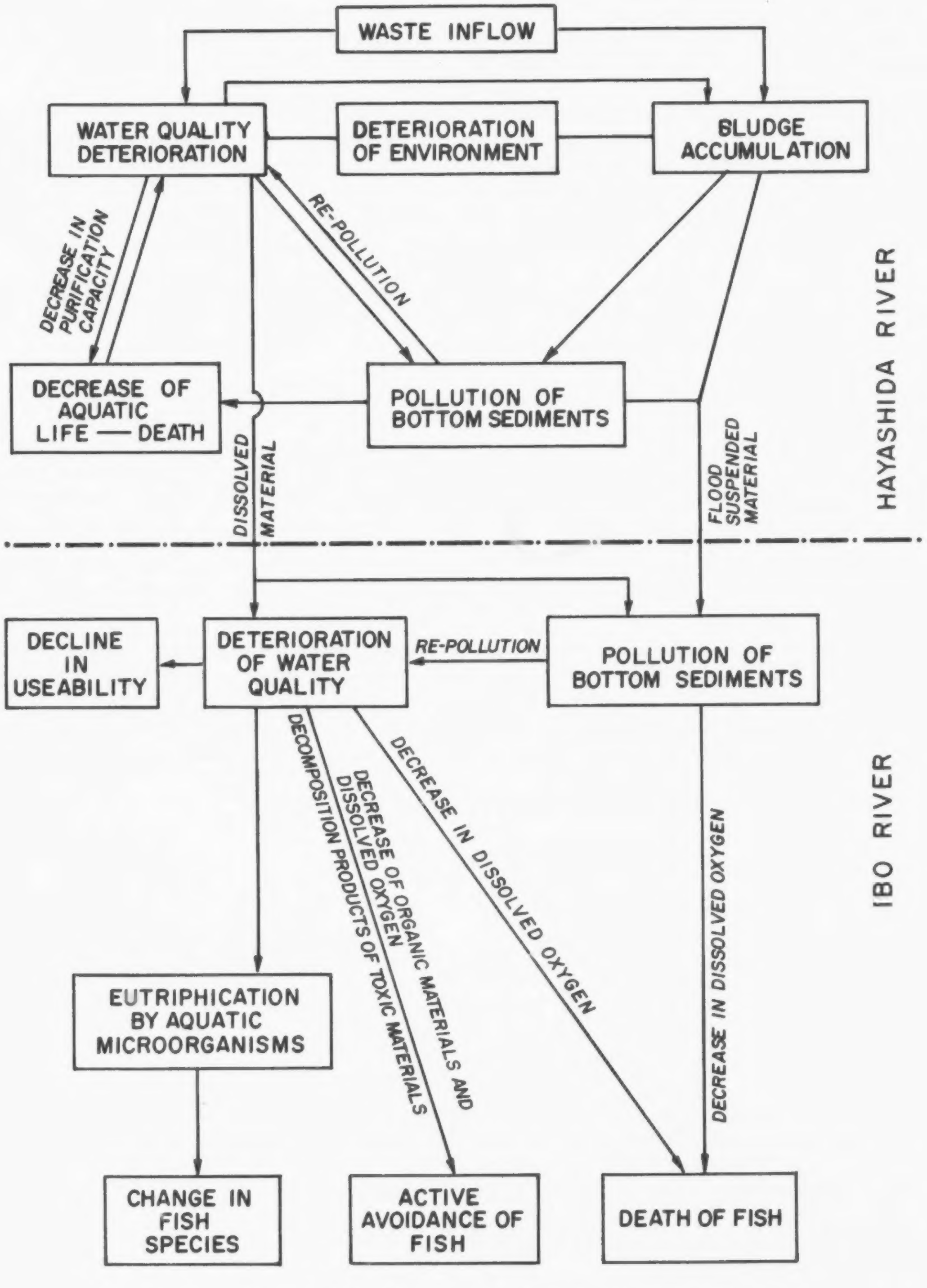

WASTE INFLOW
WATER QUALITY DETERIORATION
DETERIORATION OF ENVIRONMENT
BLUDGE ACCUMULATION
DECREASE IN PURIFICATION CAPACITY
RE-POLLUTION
DECREASE OF AQUATIC LIFE — DEATH
POLLUTION OF BOTTOM SEDIMENTS
DISSOLVED MATERIAL
FLOOD SUSPENDED MATERIAL
HAYASHIDA RIVER
DECLINE IN USEABILITY
DETERIORATION OF WATER QUALITY
RE-POLLUTION
POLLUTION OF BOTTOM SEDIMENTS
DECOMPOSITION PRODUCTS OF TOXIC MATERIALS
DECREASE OF ORGANIC MATERIALS AND DISSOLVED OXYGEN
DECREASE IN DISSOLVED OXYGEN
DECREASE IN DISSOLVED OXYGEN
IBO RIVER
EUTRIPHICATION BY AQUATIC MICROORGANISMS
CHANGE IN FISH SPECIES
ACTIVE AVOIDANCE OF FISH
DEATH OF FISH

Figure 5. Water pollution mechanism in the Ibo River.

CRITERIA FOR REMOVAL OF ORGANIC SLUDGE

Although organic sludge is detrimental to the environment there are yet no rules to judge whether river bottom dredging is necessary. However, fixed dredging criteria have been developed for major ports.

Provisional standards for removal of organic sludge have been set by Hyogo Prefecture so that dredging can be performed as a eutrophication control measure. According to these standards, the necessity for removal is based on three criteria: 1) volatile and fixed suspended matter, 2) COD and 3) sulfide. Sulfide can be omitted from the standards for rivers because little sulfide formation occurs in fresh water.

Levels of 15% volatile and fixed suspended matter and 20 mg/g COD (dry weight) are set as the standards. If one of these criteria is exceeded the bottom sediment should be removed.

The sand-gravel component does not need to be removed but almost all the sludge should be dredged, as shown in Figure 4. As a consequence, the major area to be dredged is around the confluence where the average depth of accumulated sediments is 0.8 m, the total area is 6,300 m^2, and the total volume is 5,000 m^3. Elsewhere in the Hayashida River there are about 3,000 m^3 of accumulated sediments.

DREDGING OF BOTTOM SEDIMENTS

In dredging organic sludge, secondary pollution must be avoided. In the selection and adoption of a plan for dredging, sludge treatment and disposal, individual processes must be considered as components of a total process. Figure 6 shows a total process beginning with the dredging of sludge and sediment and with final disposal.

This total process is divided into five major steps:

1) Prevention of Dispersion of Pollutants

2) Dredging and Removal

3) Treatment

4) Final Disposal

5) Environmental Monitoring

After various laboratory and field tests were conducted, integrated examination and analysis resulted in the following proposal for each step:

1) Dispersion Prevention: use of silt protector

2) Dredging and Removal: use of vacuum collector and dump truck mounted receiver tank

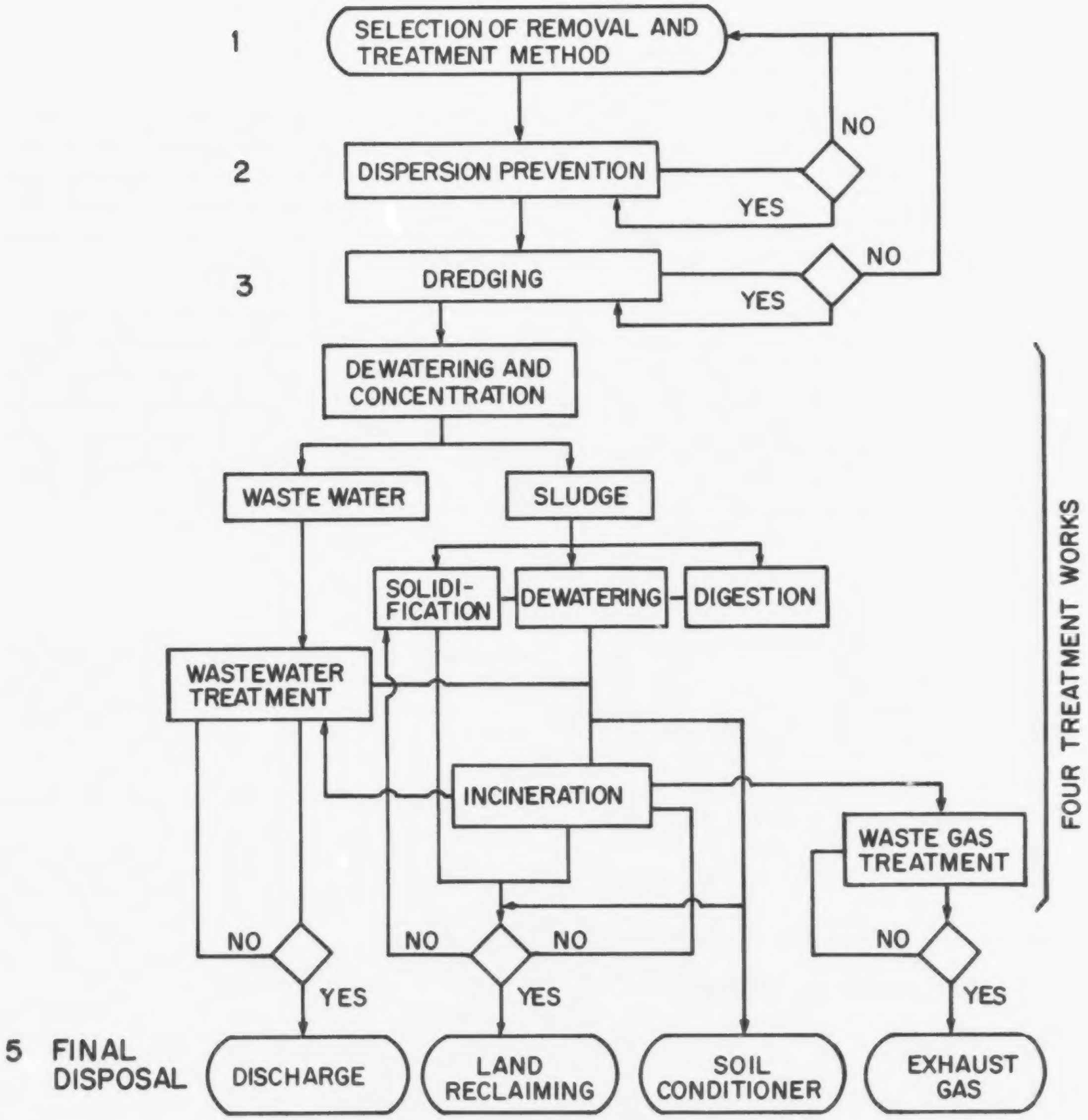

1
SELECTION OF REMOVAL AND TREATMENT METHOD
2
DISPERSION PREVENTION
NO
YES
3
DREDGING
NO
YES
DEWATERING AND CONCENTRATION
WASTE WATER
SLUDGE
SOLIDIFICATION
DEWATERING
DIGESTION
WASTEWATER TREATMENT
INCINERATION
WASTE GAS TREATMENT
FOUR TREATMENT WORKS
NO
YES
NO
NO
YES
NO
YES
5 FINAL DISPOSAL
DISCHARGE
LAND RECLAIMING
SOIL CONDITIONER
EXHAUST GAS

Figure 6. Sludge removal process.

3) Treatment: combination of natural dewatering at dry river bed and use of chemical solidifiers

4) Final Disposal: landfill disposal by dump truck

5) Environmental Monitoring: monitor water quality of dredge area and work only in winter to avoid odor problems

PREVENTION OF POLLUTANT DISPERSION

Tests were performed in winter. The "Provisional guidelines for treatment and disposal of bottom sediments containing toxic substances" provided baselines for judging water quality and bottom sediments before, during and after the dredging. Basic and supplementary monitoring points were installed beforehand. At the same time, water pollution inside and outside of the silt protector was monitored to judge the performance of the silt barrier.

One result was that the turbidity around the suction nozzle of the vacuum collector was greatly affected by water depth. When the water depth was about 10 cm, average turbidity was as high as 653 degrees, but it was about 179 degrees when the water depth was 20-40 cm. The turbidity before the work was 74-143 degrees. When water depth is shallow, extreme care must be taken. The turbidity inside and outside of the silt protector was smaller than the turbidity measured before the work at the upper and lower water layer. The silt protector should be installed in a closed loop so that currents cannot

DREDGING AND REMOVAL

In dredging organic sludge it is important to prevent secondary pollution. Disturbance of the sludge must therefore be minimized. To reduce the volume of wastewater after the dredging, the concentration of dredged material should be kept as high as possible.

Taking these factors into consideration, the dredges which could be used in the Ibo River would be the following:

1) Pump dredge (with vacuum collector)

2) Grab dredge (with closed grab bucket)

3) Bulldozer (amphibious)

4) Others

In selecting dredges, many factors such as influence on the environment, ease of use and economy were considered. The vacuum collector mounted on a truck with a receiver tank carrier for transportation of dredged sediments was adopted. In conducting tests, a 100 ps vacuum collector with a suction head having either 0.08 m x 1.0 m rectangular opening or a 0.23 m diameter round opening was used. The apparent average sludge concentration rate was in the range of 51% for the rectangular type, and 69% for the round type. When the depth of sludge accumulation is about 20 cm, the rectangular suction mouthpiece is best suited; for 40-60 cm, the round one is more effective. Soundings showed condition of the river bottom after dredging was in very good shape.

TREATMENT

Dredged sludge contains a large volume of water and its fluid nature causes problems in transportation to the dumping site. After natural dewatering using a dry river bed, chemicals are used to solidify the sludge for transporting.

Using sludge samples with water contents of 300% and 600% (dry wt. basis), solidification tests were conducted using Portland cement and fly ash. The results are shown in Figures 7 and 8. In spite of the large difference in water content, there was no significant difference in the strength per unit area. The more solidifier used and the longer the curing time, the greater the strength.

The unconfined compression strength of a cylinderically molded soil sample could not be measured. Although bottom sludge with an extremely high critical ratio was composed primarily of organics, it was relatively easy to treat.

In the field test, dredged sludge was put into a storage basin in a dry river bed, and the stages of the natural dewatering process were observed. After that, solidification tests were carried out. Table 6 and Figure 9 show the change of water content in drained sludge with time.

TABLE 6. CHANGE IN WATER CONTENT OF DREDGED SLUDGES.

Elapsed Time	Water Content
0	1,640%
3 days	1,090%
7 days	500%
14 days	370%
28 days	260%

As shown in Table 6, the natural dewatering is very effective, particularly in the first week. Although a blind culvert was installed in the sludge storage basin, most of the water permeated through the silt and a small amount came through the blind culvert.

The quality of water permeating through the blind culvert was analyzed. The turbidity was very high, and also the values of BOD and normal hexane

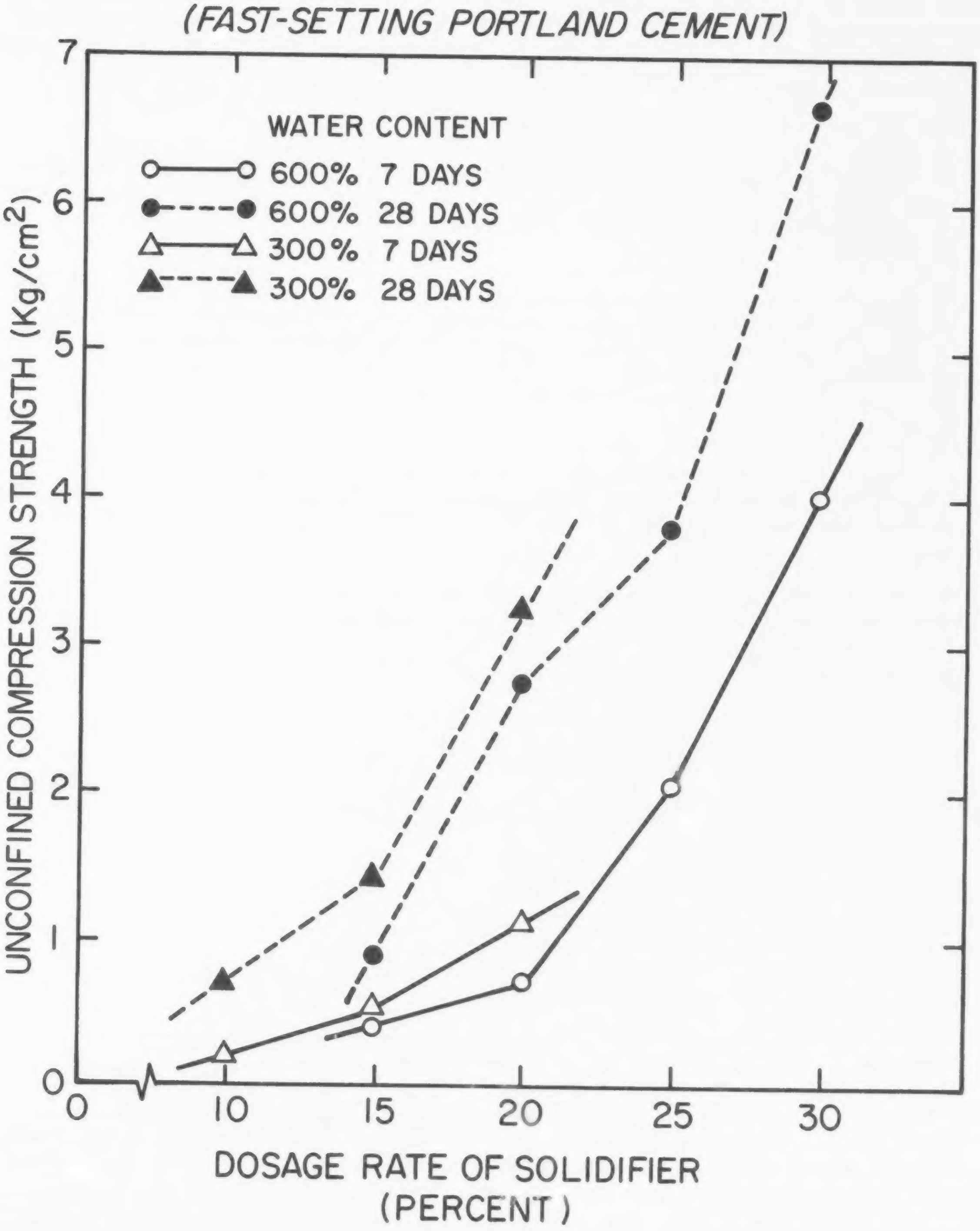


Figure 7. Relationship of solidifier dosage and strength, for curing times of 7 and 28 days.

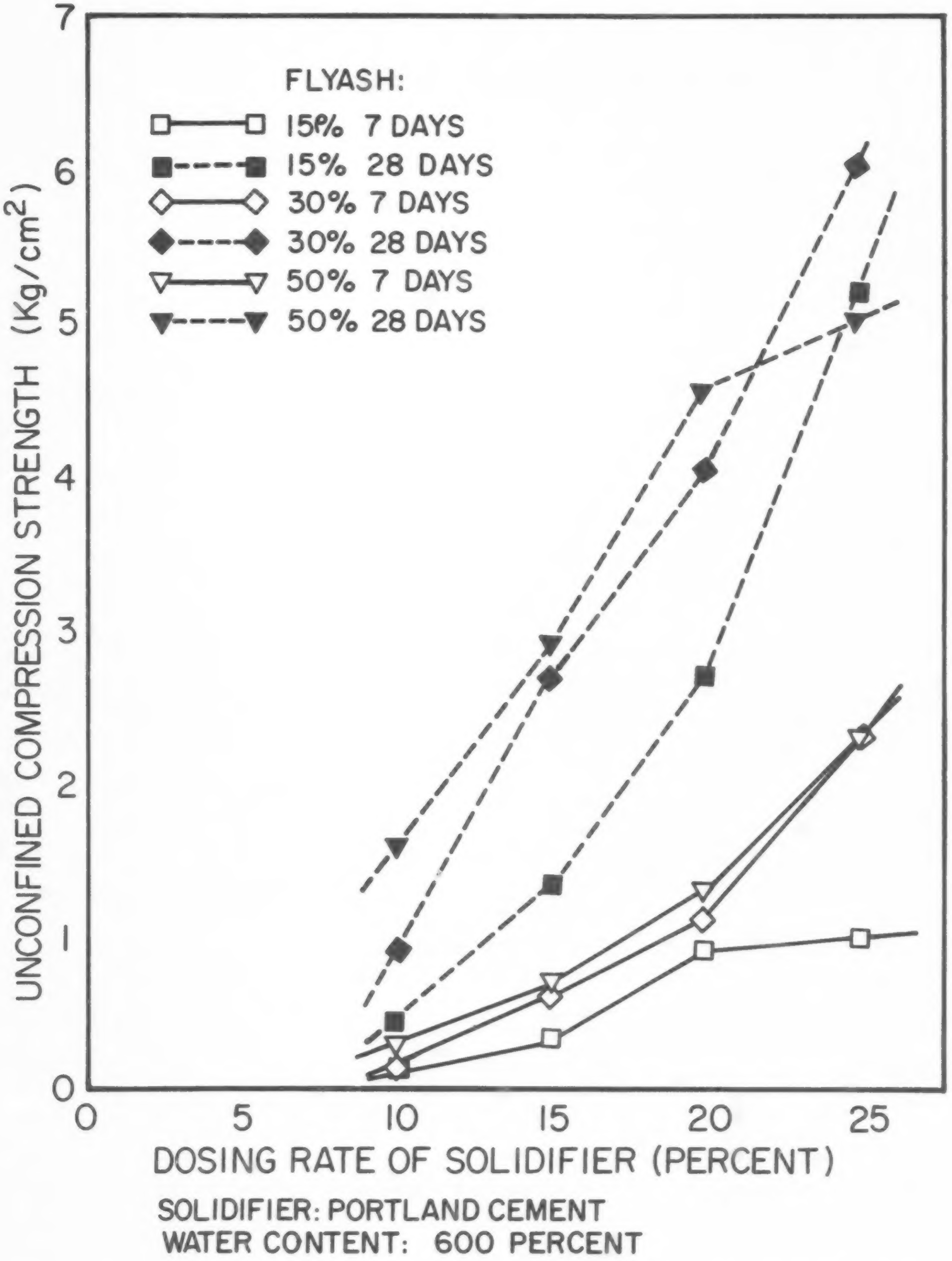


Figure 8. Relationship of dosage rate of solidifier and strength, for 7 and 28 days of curing.

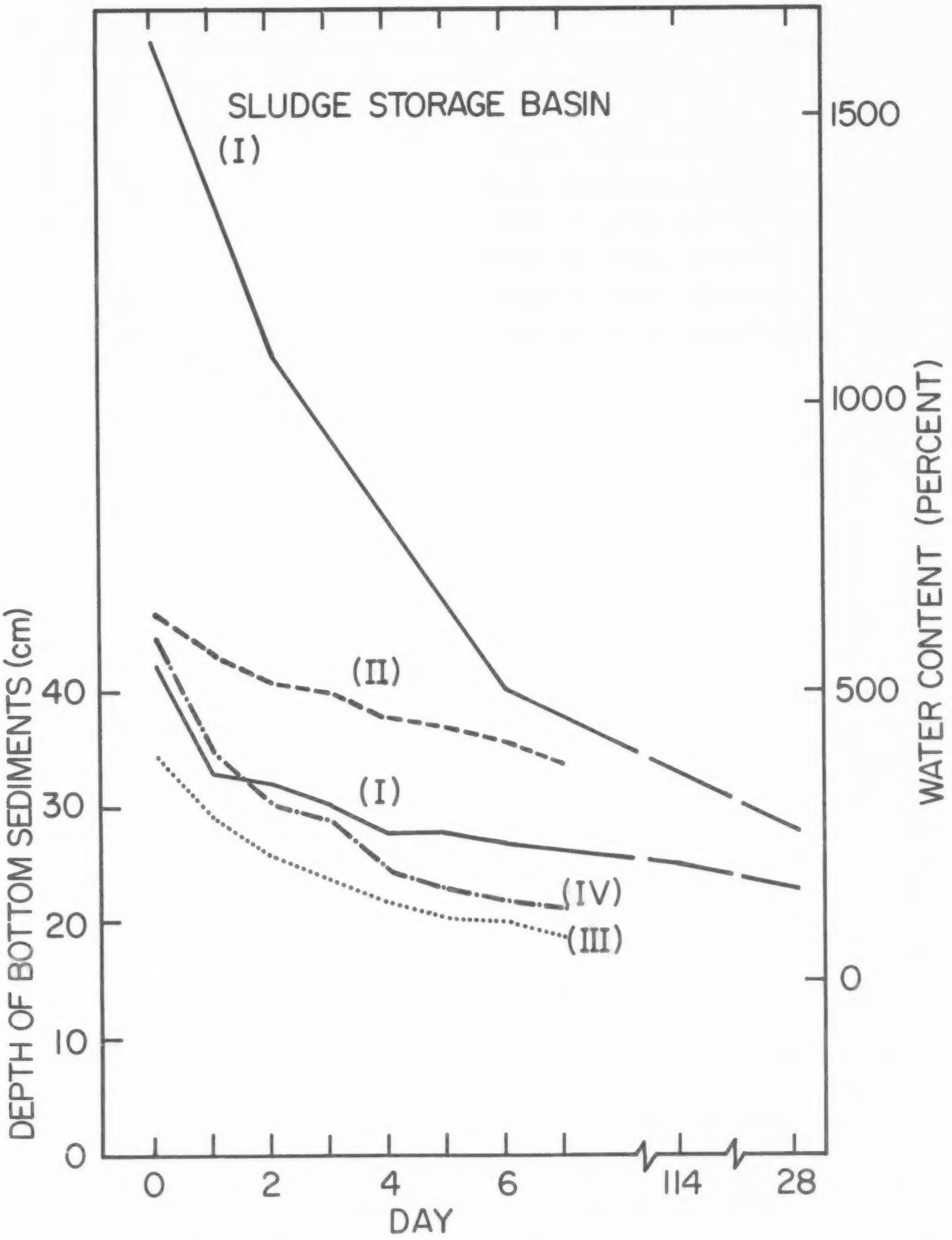


Figure 9. Change of depth of bottom sediment and water content with elapsed time.

extraction materials, but nine other tests of water quality relating to human health satisfied the discharge standards.

To obtain better water quality during operations it was necessary to adopt sand filtration for the blind culvert. Also, when the sludge storage basin is used more than twice, it has to be dug to a certain depth because of clogging.

In field tests using fast-setting Portland cement as the solidifier and fly-ash as an additive, a backhoe was used to mix the material for a period of 10 days. Unconfined compression strength using Portland cement and 20% fly-ash was as low as 0.1-0.2 kg/cm^2 for day 7, and day 28 was 0.4-0.7 kg/cm^2. In the case of 40% fly-ash day 7 was 0.2-0.3 kg/cm^2 and day 28 was 1.1 kg/cm^2 as shown in Figure 10. The critical ratio was in the range of 1.5-4.0 in the latter case. The volume was increased by about 18%, when 40% of the additive was used.

FINAL DISPOSAL

The strength needed for excavation, dumping and covering of treated sludge is 0.3 kg/cm^2 for excavation and loading, more than 1.0 kg/cm^2 for transportation and dumping, 0.8 kg/cm^2 for covering, and 1.0 kg/cm^2 for banking after reworking the soil.

Because the solidified sludge was intertwined with animal hairs that existed in the bottom sediments, the necessary strength for excavation and loading, transportation and dumping, covering, and banking were respectively 0.1-0.2 kg/cm^2, 0.4 kg/cm^2 or more, 0.3 kg/cm^2 or more, and 0.5 kg/cm^2 in terms of the strength after disturbance.

The most economical conditions that met these figures are as follows:

1) Solidifier : Fast-setting Portland cement 10%
2) Additive : fly-ash 20%
3) Curing time : not less than 28 days
4) Water content of treated sludge : not more than 70%

Ref: Above conditions are based on sludge with an initial water content of 500% and the use of a backhoe for mixing.

Before covering operations began it is necessary to wait for natural de-watering at the dumping site as well as for the recombination of soil particles, since the strength does not reach 0.3 kg/cm^2 at the time of transportation. From the test data on the critical ratio, solidified sludge cannot be used for banking materials. It cannot be used as backfill material in revetment, for the same reason.

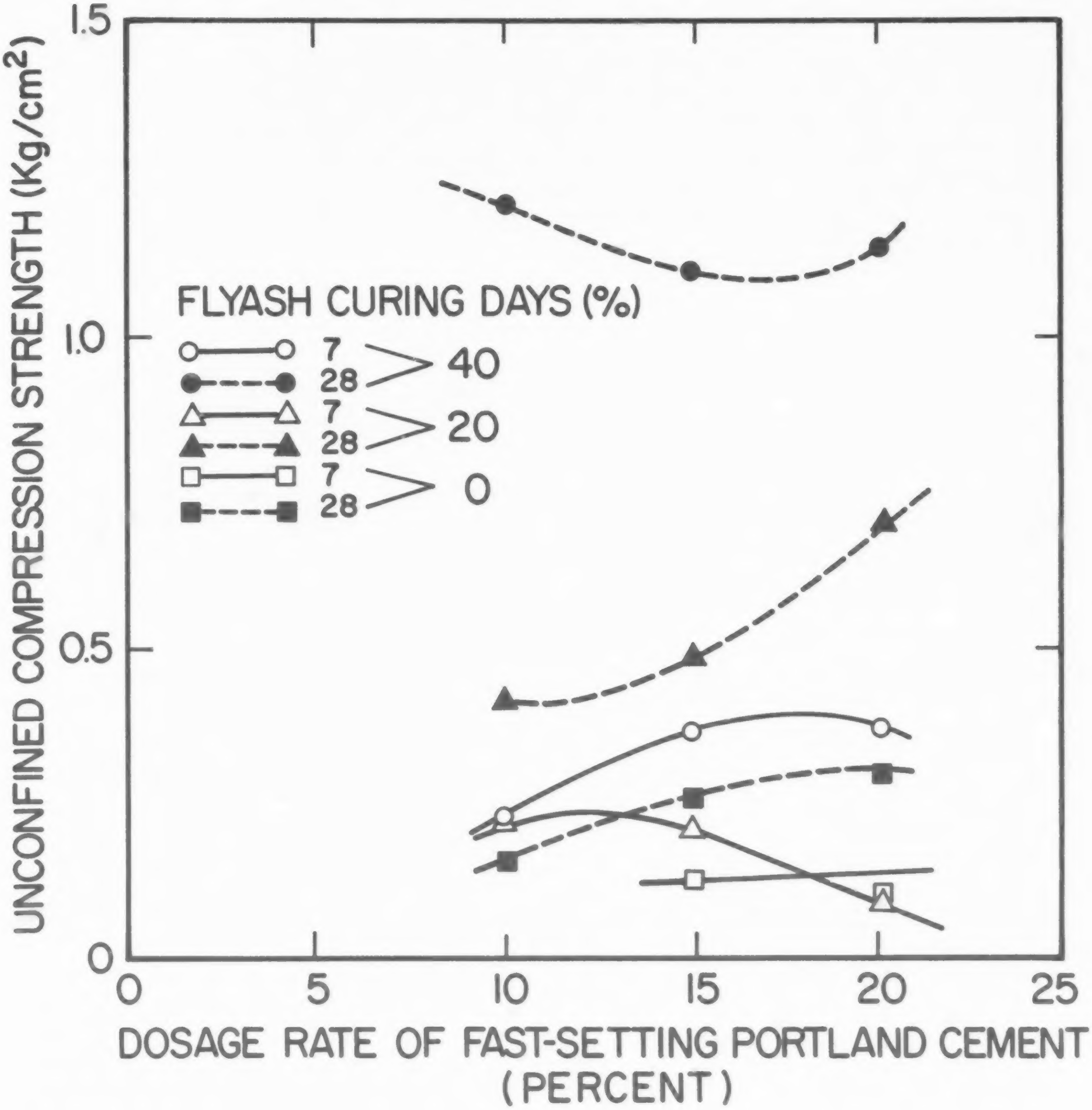


Figure 10. Unconfined compression strength and dosage rate of solidifier for 7 and 28 days of curing.

Consequently, dumping in dry river beds or transportating them to a neighboring land reclamation site and covering the surface with better soil may be the only possible way of final disposal.

ENVIRONMENTAL MONITORING

Downstream of the dredging site are industrial water intakes and a commercial fishery. Careful monitoring is necessary to avoid the generation of secondary pollution by dredging which would have a negative impact on these water users. As mentioned earlier, the water quality has been monitored for several factors such as turbidity, pH, BOD, COD, normal hexane extraction and DO at basic and supplementary monitoring points installed downstream from the worksite. Downstream samples are compared with control samples to prevent water quality deterioration due to dredging. The installation of the closed silt protector proved to forestall such problems.

Odor monitoring is also required. During the field tests in winter neither H_2S nor methyl mercaptan were detected at a distance of 5 to 50 cm from the surface of dredged sludge in the disposal basin. When heated inside to 30¼C, 5 mg/l H_2S was detected. A threshold concentration for human health is 10-15 mg/l. At this level, H_2S will cause eye irritation within 6 hours.

CONCLUSION

The particle size distribution curve for bottom sediments before and after the test dredging is shown in Figure 11. The bottom sediments prior to dredging are composed mainly of minute particles; after dredging they consist more of coarse particles.

The cost for dredging depends on site and volume. A rough estimate is about 8,700-10,000 yen per cubic meter, of which the solidification treatment costs 3,700 yen.

In conclusion, after extensive laboratory and field tests, useful knowledge has been obtained which is applicable to actual dredging.

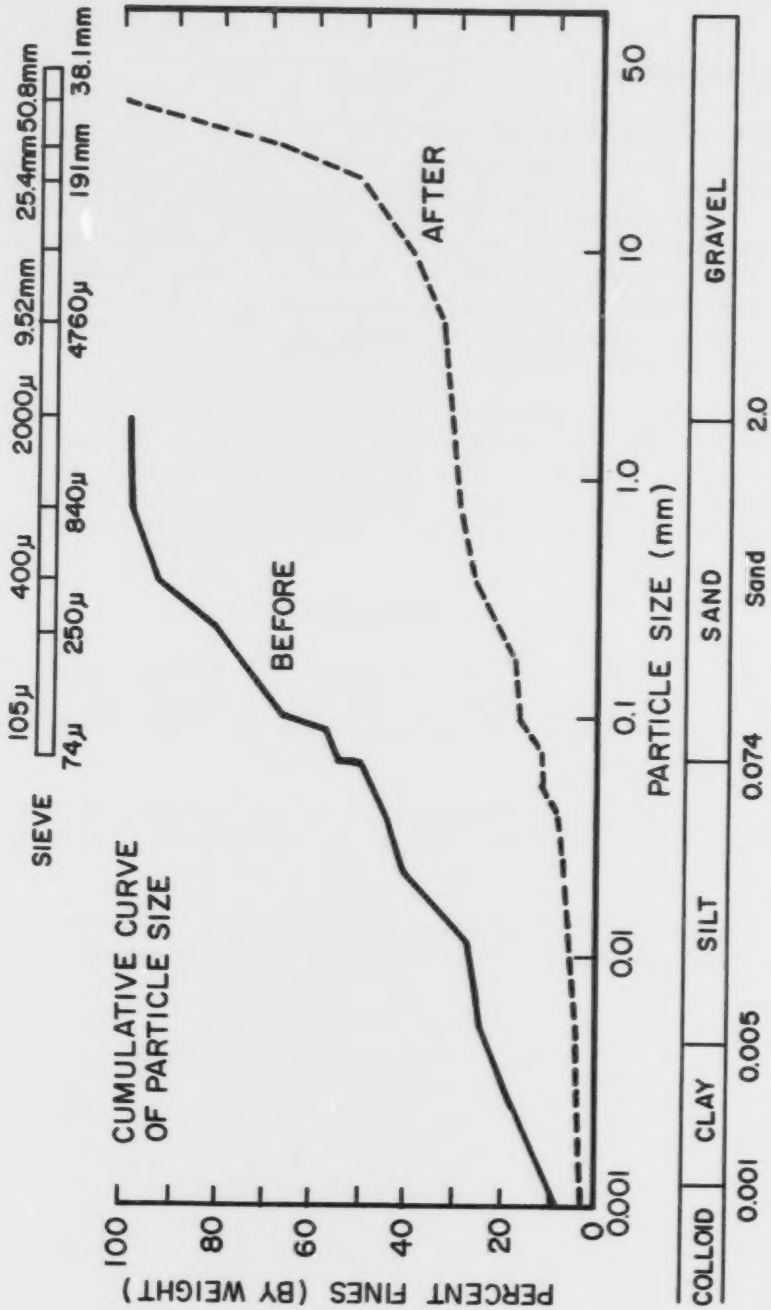


Figure 11. Particle distribution curve for bottom sediments before and after dredging.

POLLUTION CONTROL IN TOKYO BAY

Masai Yako and Keitchi Akimotc
2nd District Port Construction Bureau
Ministry of Transport, Japan

INTRODUCTION

This report indicates the distribution and properties of the sapropel which has accumulated in Tokyo Bay, clarifies the effects of this sapropel on water quality and the biota and investigates the results of dredging designed to control pollution from this source.

POLLUTION IN TOKYO BAY

BOTTOM DEPOSITS

Sapropel is a soft mud, containing large amounts of organic detritus such as algae, pollen, spores or animals. Sapropel accumulates under reducing conditions (anaerobic areas) in calm sea areas. The sapropel was found to be widely distributed from off Haneda to the middle of the bay with a thickness of 40-50 cm (Figure 1). Samples of sapropel and the underlying mud, 50-80 cm thick, were collected together without disturbing the layers and the samples were analyzed.

The conditions of the bottom deposits were classified according to 1) surface layer, 10 cm down from the top of the sapropel, and 2) substratum, the clay and silty layer 50-60 cm below the sapropel (Figure 2).

Analyses of the deposits showed that the surface layer had higher values than the substratum with respect to water content, sulfides, ignition loss and chemical oxygen demand (COD). There was also a remarkable amount of organic pollution in the surface layer in the innermost part of the bay. In addition to these organic pollution indices, the nutrient content (nitrogen and phosphorus) was high and matched the areas where the sapropel was thick (Figures 3-8).

WATER QUALITY

In summer there was a thermocline and halocline at depths of 5-10 m. Water quality differed remarkably above and below this layer. pH was high in the surface layer and tended to be low in the bottom layer. Dissolved oxygen (DO) was supersaturated in the surface layer, while the bottom water in the innermost and middle parts of the bay was poor in oxygen. COD and total

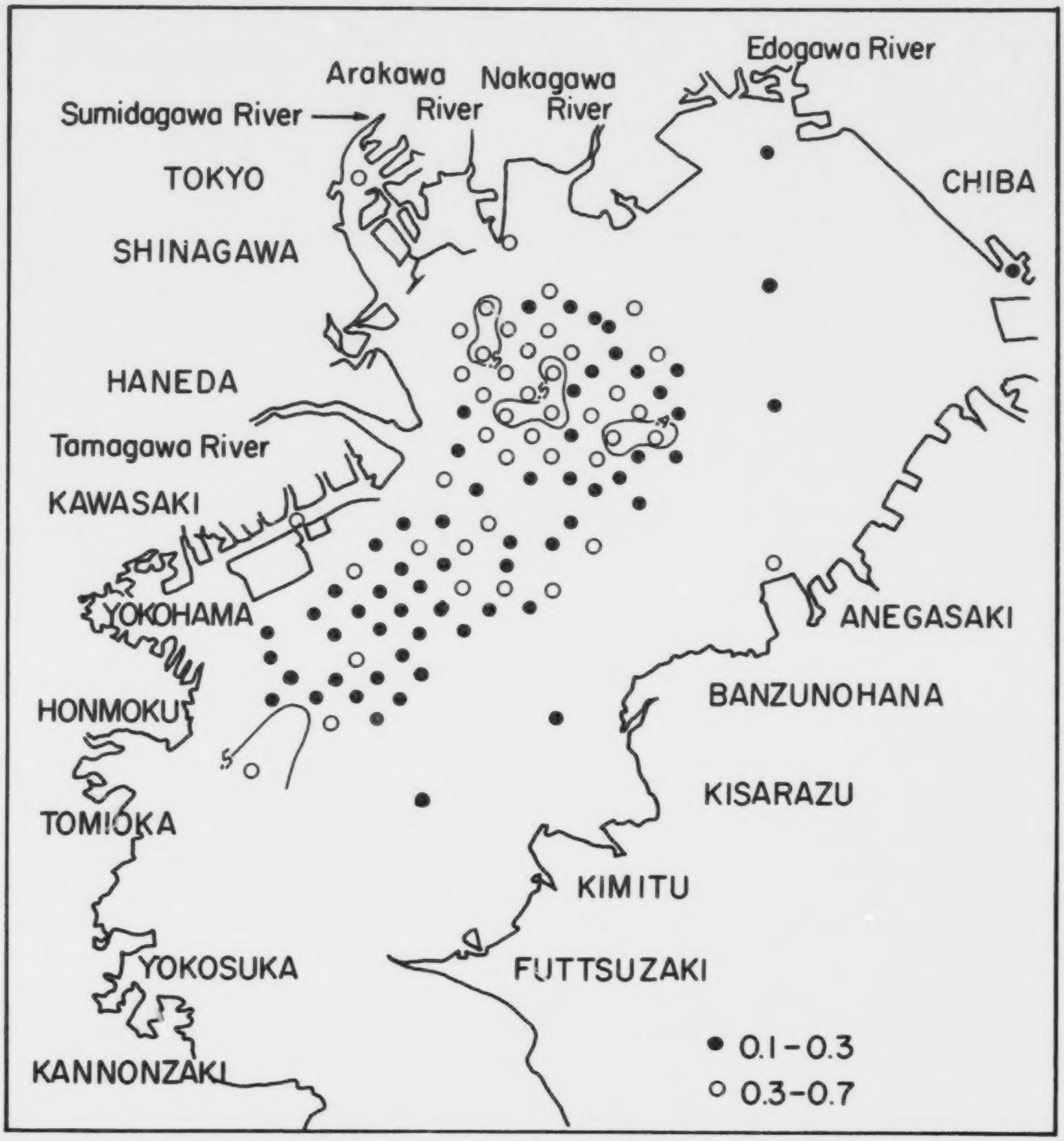

Edogawa River
Arakawa River
Nakagawa River
Sumidagawa River
TOKYO
CHIBA
SHINAGAWA
HANEDA
Tamagawa River
KAWASAKI
YOKOHAMA
ANEGASAKI
BANZUNOHANA
HONMOKU
KISARAZU
TOMIOKA
KIMITU
YOKOSUKA
FUTTSUZAKI
KANNONZAKI
● 0.1-0.3
○ 0.3-0.7

Figure 1. Distribution and thickness of sapropel in Tokyo Bay (in meters).

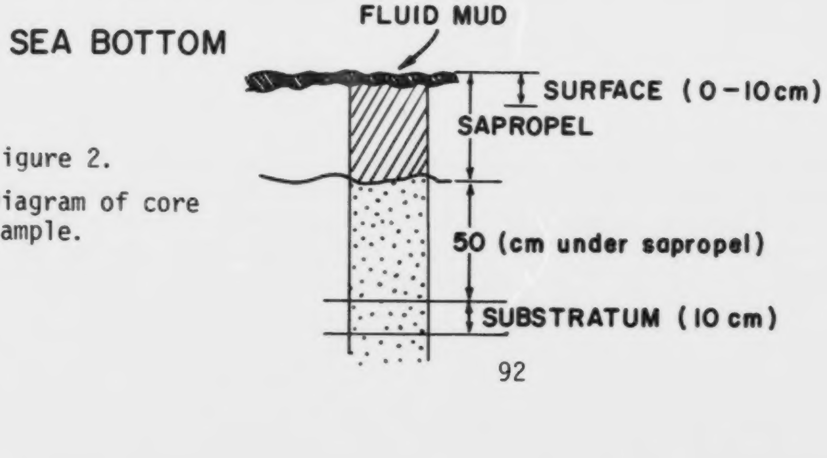

SEA BOTTOM
FLUID MUD
SURFACE (0-10cm)
SAPROPEL
50 (cm under sapropel)
SUBSTRATUM (10 cm)

Figure 2.

Diagram of core sample.

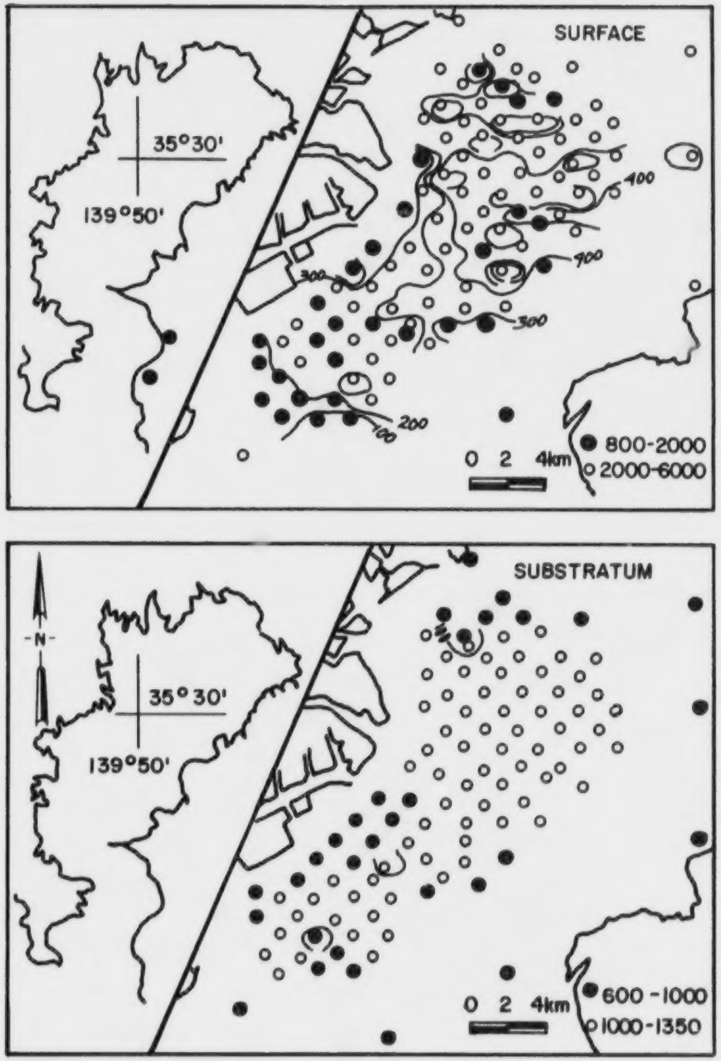

SURFACE
35°30'
139°50'
400
900
300
200
100
0 2 4km
● 800-2000
○ 2000-6000
SUBSTRATUM
N
35°30'
139°50'
0 2 4km
● 600-1000
○ 1000-1350

Figure 3. Water content (%) September 1976.

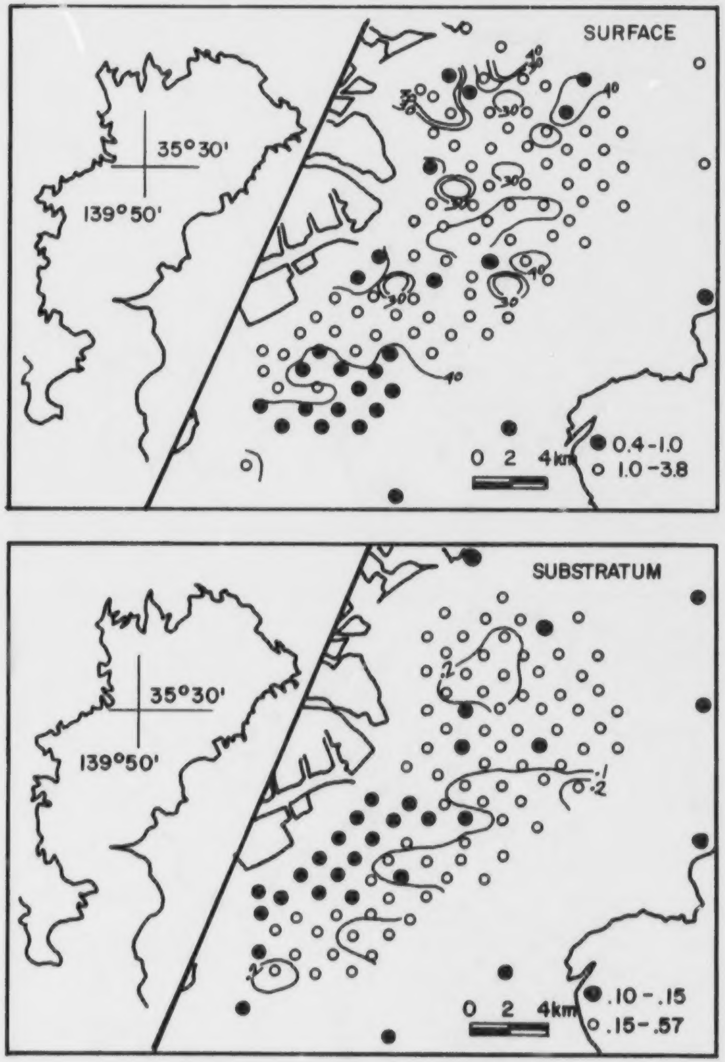

SURFACE
35°30'
139°50'
10
30
0 2 4km
● 0.4-1.0
○ 1.0-3.8
SUBSTRATUM
35°30'
139°50'
.1
.2
0 2 4km
● .10-.15
○ .15-.57

Figure 4. Sulfides (mg/g) September 1976.

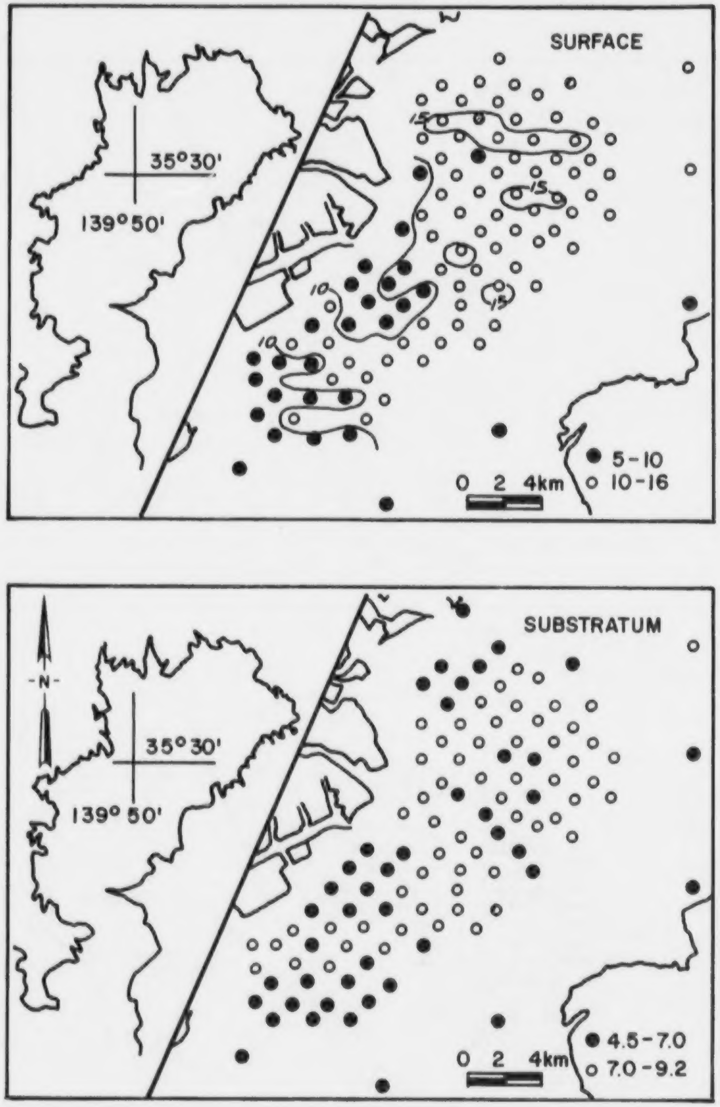

SURFACE
35°30'
139°50'
15
10
● 5-10
○ 10-16
0 2 4km
SUBSTRATUM
N
● 4.5-7.0
○ 7.0-9.2

Figure 5. Ignition loss (%) September 1976.

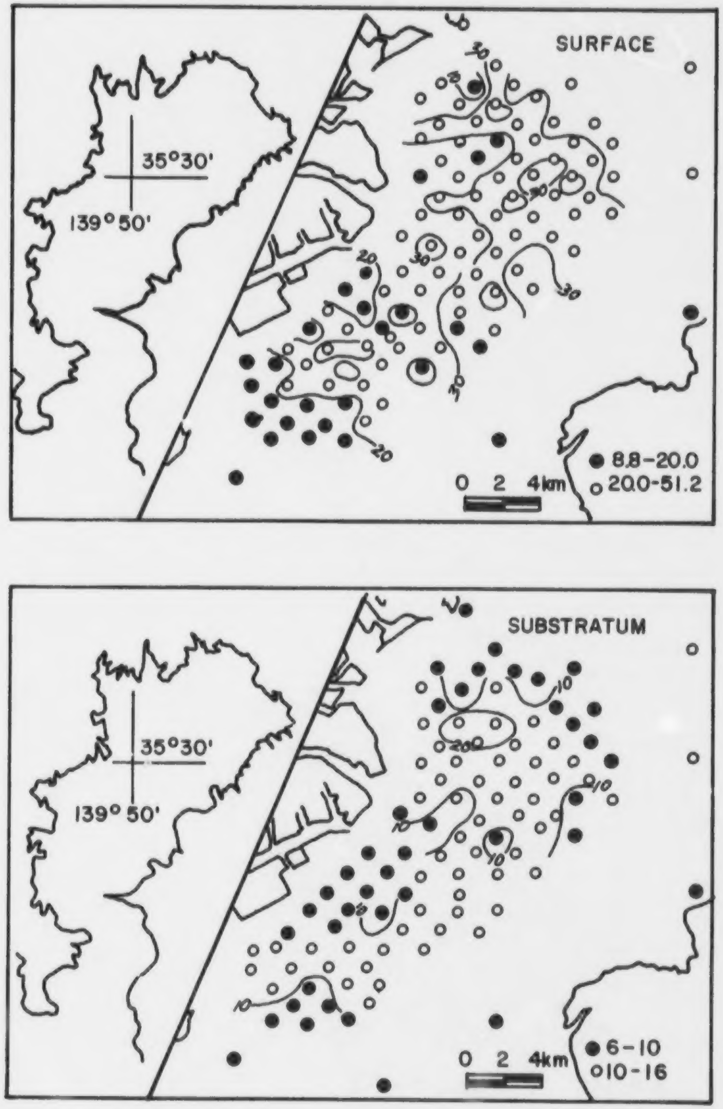

SURFACE
35°30'
139°50'
30
20
● 8.8-20.0
○ 20.0-51.2
0 2 4km
SUBSTRATUM
20
10
● 6-10
○ 10-16

Figure 6. COD (mg/g) September 1976.

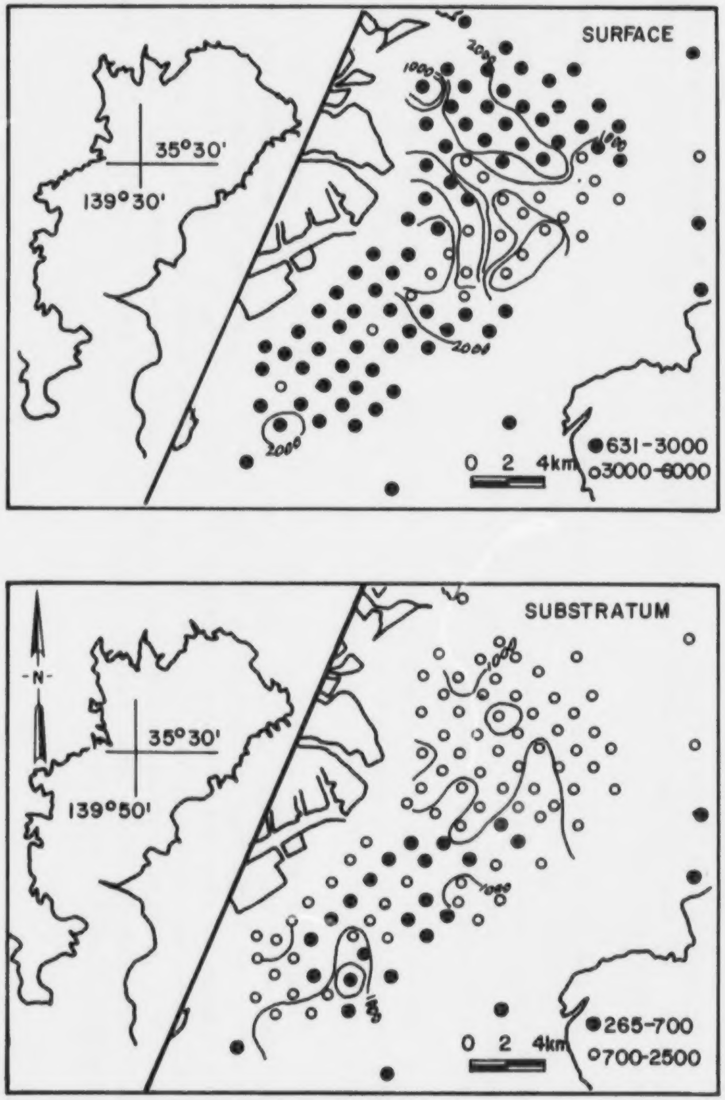

SURFACE
35°30'
139°30'
1000
2000
1000
2000
2000
0 2 4km
●631-3000
○3000-6000
SUBSTRATUM
-N-
35°30'
139°50'
1000
1000
0 2 4km
●265-700
○700-2500

Figure 7. T-N (mg/kg) September 1976.

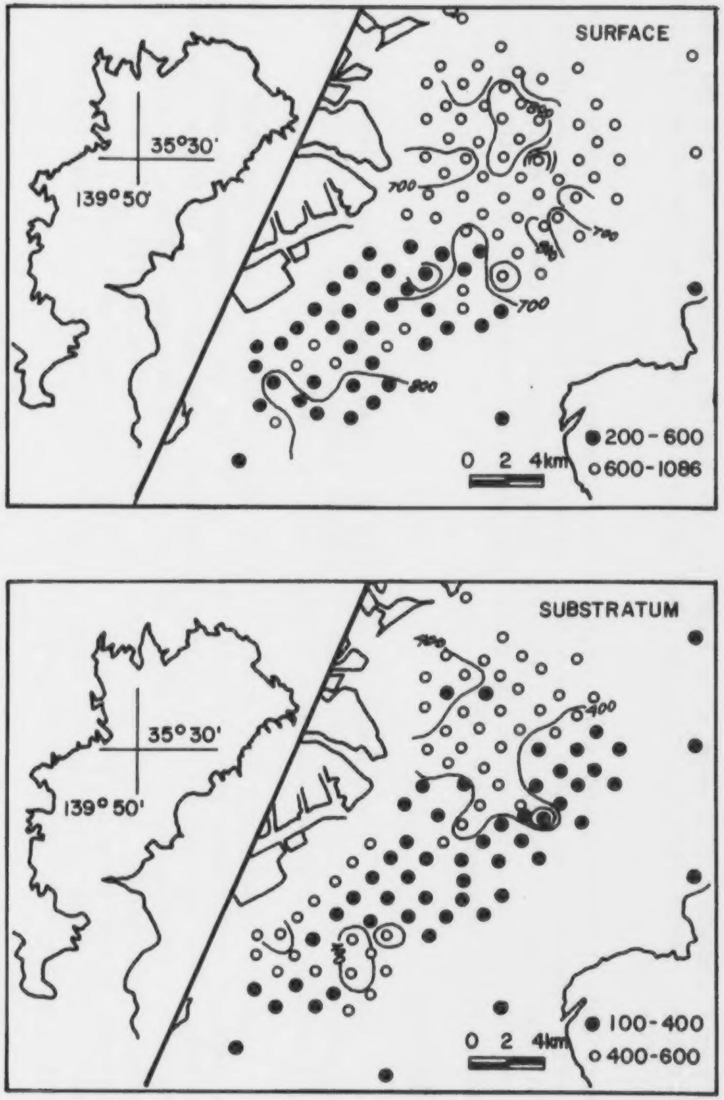

SURFACE
35°30'
139°50'
700
700
700
800
0 2 4km
●200-600
○600-1086
SUBSTRATUM
35°30'
139°50'
700
400
0 2 4km
●100-400
○400-600

Figure 8. T-P (mg/kg) September 1976.

organic carbon (TOC) were also high in the surface layer and low in the substratum. These results indicate that above the thermocline the water is rich in organics.

Nutrient concentrations were high in the coastal regions but tended to decrease towards the mouth of the bay. There were high values for phosphorus, in the form of phosphates (PO_4-P), and nitrogen in the form of ammonia (NH_4-N), in the water directly above the sea bed, corresponding to the zone of oxygen-poor water (Figures 9-11).

BENTHOS

Sampling was performed twice with a Koken mud sampler. This device samples an area of 1/15 m^2 with a sample size of 5.6 l. The macrobenthos retained on 1 mm-mesh sieve was identified and counted.

In the summer survey, areas totally devoid of life were observed in the innermost and middle parts of the bay. Only one or two organisms were found nearby. The species found included polychaetes and mollusks in the middle and mouth of the bay and crustaceans at the mouth of the bay. There tended to be greater diversity toward the mouth of the bay (Figure 11). In the winter survey, there were no areas devoid of life, but there were many places in the inner bay where only two or three organisms were found. This was correlated with the spread of organic pollution.

Figure 12 shows the relationship between the benthos and the values for COD, sulfides and ignition loss in the mud deposits of Tokyo Bay.

According to these results, polychaetes gradually decreased when the value of COD fell below 15 mg/g, sulfides fell below 0.5 mg/g and the ignition loss fell below 10%.

In the case of crustaceans, maximum numbers were seen at a COD of about 13 mg/g, sulfide content of 0.25 mg/g and an ignition loss of about 7.5%. When these values became higher or lower, the number of crustaceans decreased as well as percent contribution to the population. This suggests that there are optimum values for crustacean success.

The total number of benthic organisms, the number of species and the biotic index (BI) showed optimum values for a COD of 8-10 mg/g, sulfide content of 0.3-0.5 mg/g and an ignition loss of about 5%.

Figure 13 shows a histogram correlating the number of species in the benthos with the values of COD, sulfide and ignition loss and mud deposits in a survey of Tokyo Bay conducted in October, 1972. According to these results, the number of species reached a maximum at a COD of 5 mg/g, sulfide content of 0.25 mg/g and an ignition loss of about 5%. When these results are compared with the present surveys, the COD value is seen to be rather low, but the sulfides and ignition loss values match closely the best conditions from the standpoint of the benthos.

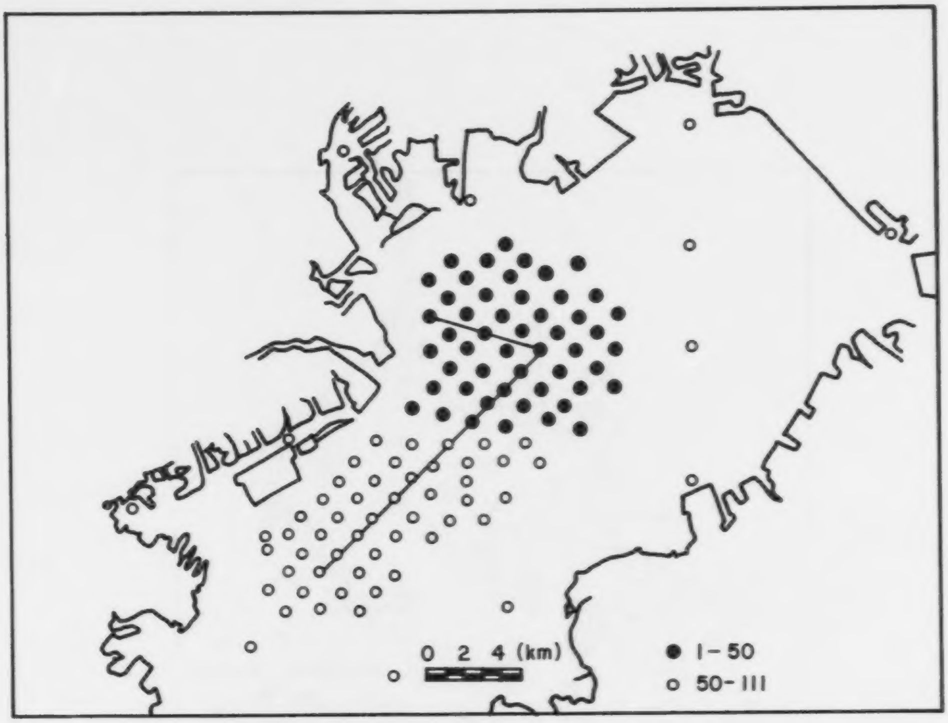

0 2 4 (km)
● 1-50
○ 50-111

Figure 9. Sampling stations for water quality gradients.

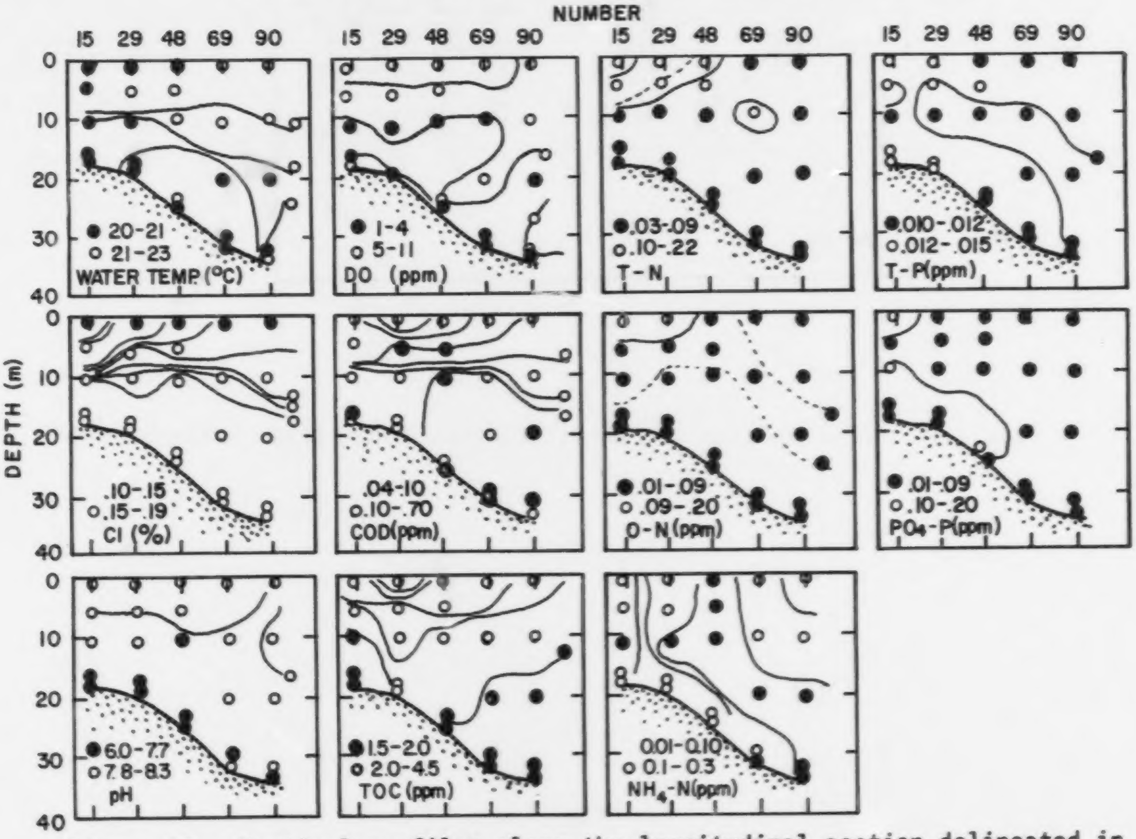


Figure 10. Vertical profiles along the longitudinal section delineated in Figure 9.

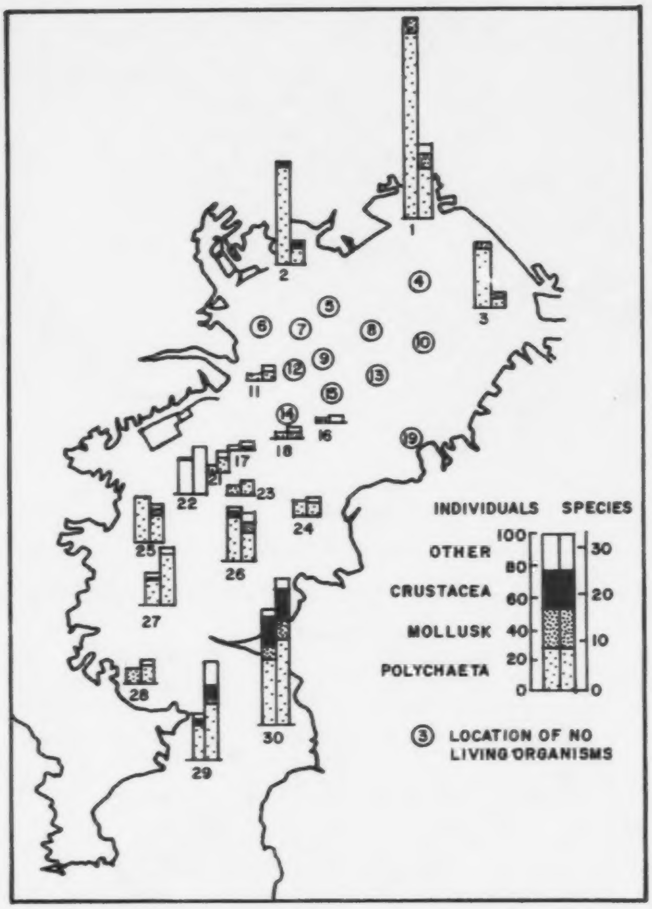

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
21
22
23
24
25
26
27
28
29
30
INDIVIDUALS
SPECIES
OTHER
CRUSTACEA
MOLLUSK
POLYCHAETA
100
80
60
40
20
0
30
20
10
0
③ LOCATION OF NO LIVING ORGANISMS

Figure 11. Distribution of benthic species composition September 1977.

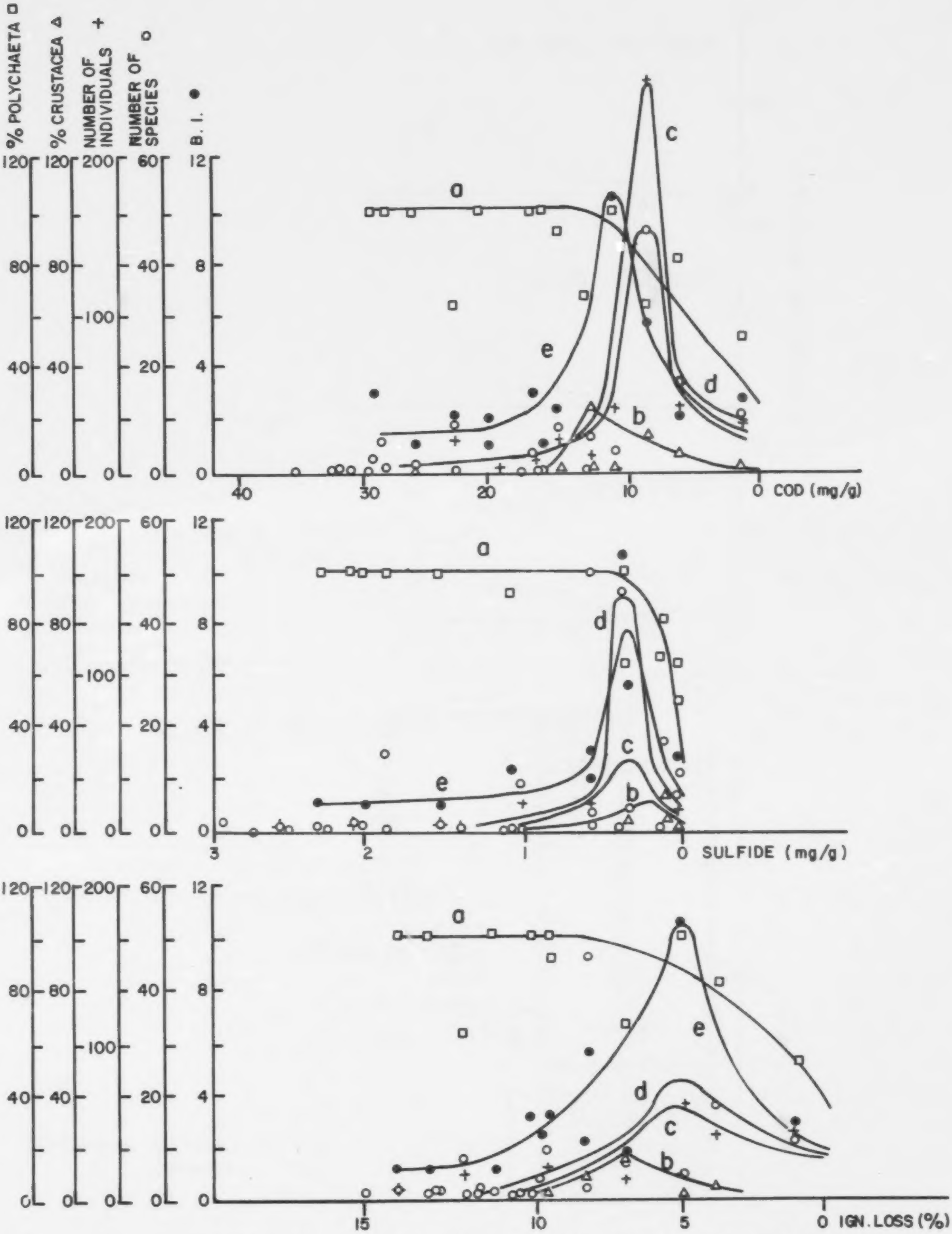


Figure 12. Correlation between zoobenthos and COD, sulfide and ignition loss.

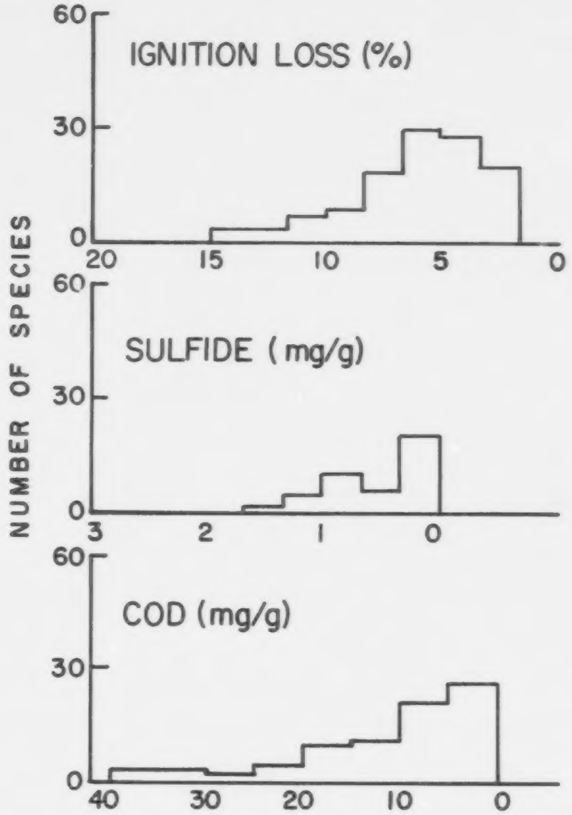


Figure 13. Histogram correlating benthic species with sediment values.

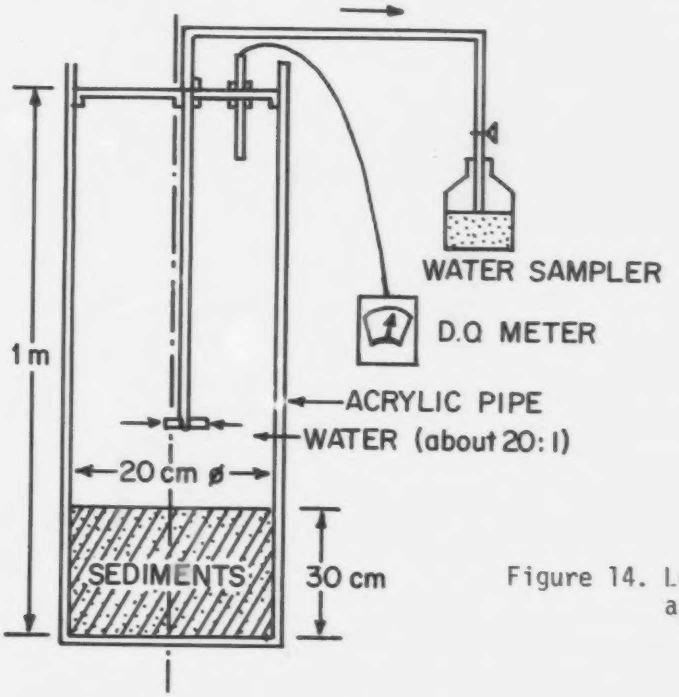

WATER SAMPLER
D.O METER
1 m
ACRYLIC PIPE
WATER (about 20:1)
20 cm ø
SEDIMENTS
30 cm

Figure 14. Leaching experiment apparatus.

EFFECT OF BOTTOM DEPOSITS ON WATER QUALITY

There are two ways in which bottom deposits affect water quality - by leaching of organic matter and nutrients from the deposits, and by the consumption of dissolved oxygen from the water.

In Japan there have been few experiments on the amounts of substances leached into the sea, and there is only a general notion of the way the contents and temperature of organic matter and nutrient salts in the mud, and dissolved oxygen in the water just above the sea bottom, influence the leaching process.

There are often cases where deposits consume the dissolved oxygen in the water and the oxygen-poor conditions which produce a cyclic pattern promoting further deterioration of the deposits, lower water quality, and adverse effects on the benthos.

In the leaching experiments, samples were taken simultaneously from the surface layer of the deposit (1-30 cm) and the substratum (50-80 cm under the sapropel). The water directly above the bottom was also taken for use in the experiment. Figure 14 shows the experimental apparatus.

The experiment was performed for five days and the amount leached per unit area per day ($mg/m^2/day$) was calculated from changes in the concentration in the water. The average amount leached was the average of these daily values.

The average values obtained in experiments performed in September, 1976, and September, 1977, are shown in Table 1. Large differences were observed between the surface layer and the substratum.

TABLE 1. AVERAGE AMOUNTS LEACHED ($mg/m^2/day$)

	COD	TOC	T-N	T-P
Surface layer	285	192	145	27
Substratum	232	125	136	10

OXYGEN CONSUMPTION RATE IN THE DEPOSITS

The samples and *in situ* seawater for this experiment were collected by the same method as used in the leaching experiment. After they were brought to the laboratory, they were reaerated in a constant temperature chamber (20 ± 2°C). The experiment was started when the DO concentration in the seawater increased. The DO and S^- concentrations in this water were measured at intervals.

The experiment was performed for five days and the average oxygen consumption was calculated using the same method as in the leaching experiment. The $S^{=}$ ion was converted to oxygen in the ratio of 2 moles of O_2 produced to 1 mole of $S^{=}$, based on the equation $SO_4^{=} \rightleftarrows S^{=} + 2O_2$ (gas).

The results gave an average oxygen consumption rate of 2.5g O_2/m^2/day in the surface layer and 1.32g O_2/m^2/day in the substratum. The surface layer consumed 1.7 times more oxygen than the substratum. These results show that the sapropel has a deleterious effect on the aquatic environment and that the environment can be restored by removing the sapropel.

THE RESULTS OF REMOVING THE SAPROPEL

Seawater purification can be broken into two types - biochemical purification produced by microorganisms, or physical purification such as seawater exchange.

The use of microorganisms involves their ingestion of organic matter present in the water. Part of this matter is converted to substances necessary for the growth of the microorganism and the remainder becomes a respiration byproduct which is oxidized to inorganic material. The bacteria eventually die and this material is utilized by other bacteria. There is, therefore, a partial change to inorganic material in this process. The organic matter is decomposed more quickly as the number of times the organic matter passes through the bacteria increases. When there are protozoa or zooplankton present to feed upon these bacteria, the number of cycles through organisms increases and the organic matter is decomposed at a faster rate.

This ingestion of organic matter by microorganisms is part of the process by which substances circulate in the water. The way to understand the process of biological control of organic matter is to investigate the circulation of organic matter in the biota and the cyclic rate. Figure 15 shows the circulation of organic material in Tokyo Bay.

Because this survey was conducted in summer when there is a lot of turbidity, many points remain unclear. Various coefficients of the material circulation model were determined by site observations and laboratory experiments. The idea was to determine, by means of numerical simulation, what degree of water purification can be expected by the removal of the sapropel.

BASIC EQUATIONS OF THE EUTROPHICATION MODEL

In the preparation of the eutrophication simulation model, the following five items had to be quantified to simulate the seawater turbidity mechanism:

1) The quantities of COD and nutrient salts in the water which flows into the bay

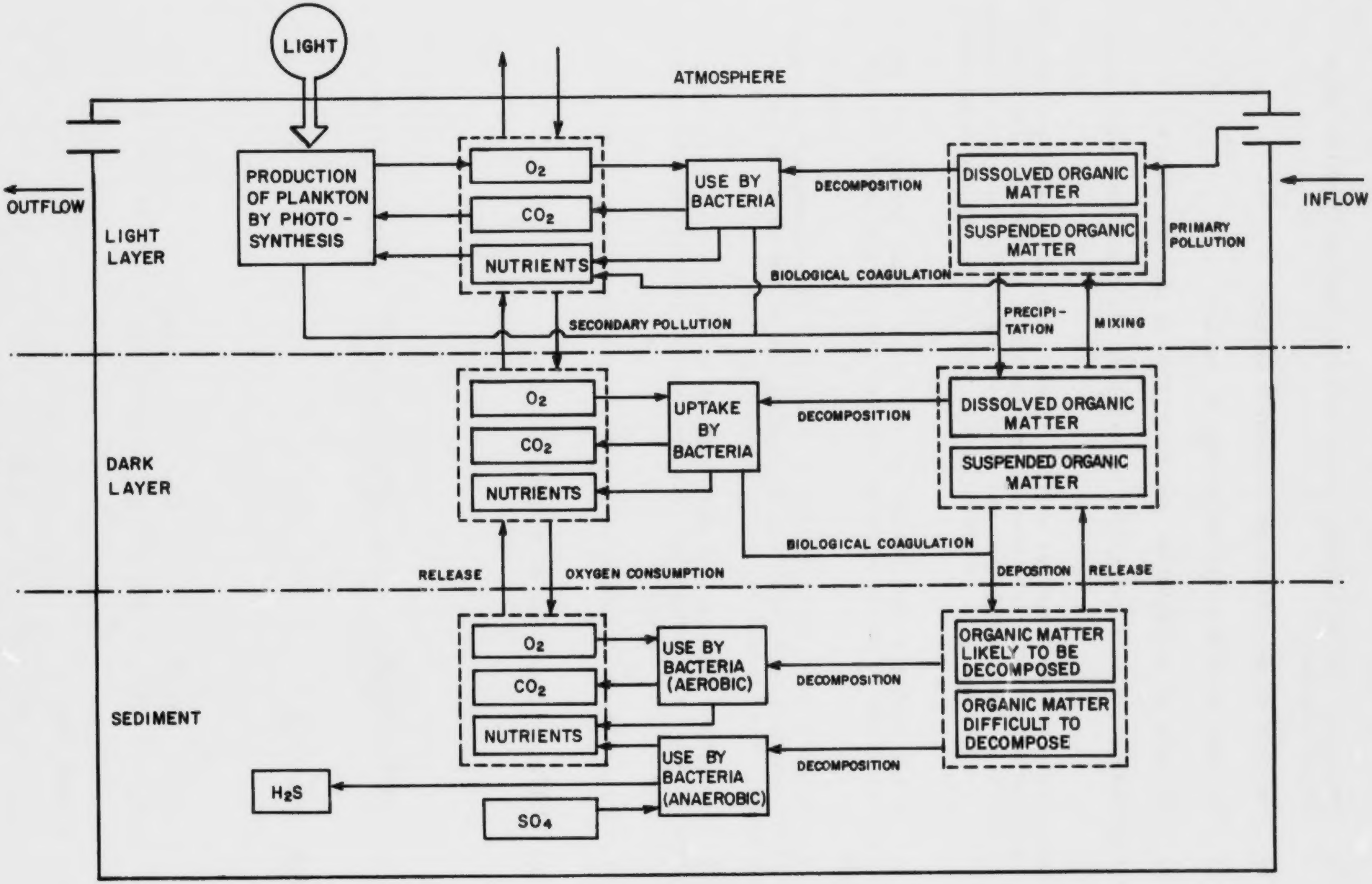

LIGHT
ATMOSPHERE
OUTFLOW
INFLOW
LIGHT LAYER
PRODUCTION OF PLANKTON BY PHOTO - SYNTHESIS
O₂
CO₂
NUTRIENTS
USE BY BACTERIA
DECOMPOSITION
DISSOLVED ORGANIC MATTER
SUSPENDED ORGANIC MATTER
PRIMARY POLLUTION
BIOLOGICAL COAGULATION
SECONDARY POLLUTION
PRECIPI - TATION
MIXING
DARK LAYER
O₂
CO₂
NUTRIENTS
UPTAKE BY BACTERIA
DECOMPOSITION
DISSOLVED ORGANIC MATTER
SUSPENDED ORGANIC MATTER
BIOLOGICAL COAGULATION
RELEASE
OXYGEN CONSUMPTION
DEPOSITION
RELEASE
SEDIMENT
O₂
CO₂
NUTRIENTS
USE BY BACTERIA (AEROBIC)
DECOMPOSITION
ORGANIC MATTER LIKELY TO BE DECOMPOSED
ORGANIC MATTER DIFFICULT TO DECOMPOSE
USE BY BACTERIA (ANAEROBIC)
DECOMPOSITION
H₂S
SO₄

Figure 15. Process of accumulation of nutrients and organic matter.

2) The quantities of organic matter and nutrient salts which leach from the bottom mud

3) The rate of decomposition of organic matter

4) The rate of precipitation of organic matter

5) The rate of assimilation and absorption by photosynthesis

These five parameters were quantified by means of field and laboratory experiments.

Four substances were used as indices in the numerical analysis: COD, DO, limiting organic nutrients (phosphorus) and inorganic nutrients. The circulation process of these substances was expressed by developing eight diffusion equations, given below. The following points were guidelines for development of the equations:

1) In the euphotic zone, there are diffusion, flow, production, decomposition, precipitation and an exchange between the surface layer and substratum.

2) In the aphotic zone, there are diffusion, flow decomposition, precipitation, leaching from deposits, DO consumption by the deposits and exchanges between the surface layer and substratum.

3) Production involves the assimilation of inorganic nutrient salts by phytoplankton and their conversion into organic nutrient salts. This results in an increase in the COD and DO.

4) The inflow load refers only to flux into the euphotic zone.

(A) Diffusion Equations of Nutrients

1) Euphotic zone

$$\frac{\partial Noh}{\partial t} = - \frac{\partial Novh}{\partial x} - \frac{\partial Novh}{\partial y} + \frac{\partial}{\partial x}\left(Kh \frac{\partial No}{\partial x}\right) + \frac{\partial}{\partial y}\left(Kh \frac{\partial No}{\partial y}\right) - Kz(No - N'o) + \alpha N_T h$$

$$- \beta Noh - \gamma No + LNo \qquad (1)$$

$$\frac{\partial N_I h}{\partial t} = - \frac{\partial N_I h}{\partial x} - \frac{\partial N_I vh}{\partial y} + \frac{\partial}{\partial x}\left(Kh \frac{\partial N_I}{\partial x}\right) + \frac{\partial}{\partial y}\left(Kh \frac{\partial N_I}{\partial y}\right) - Kz(N_I - N'I)$$

$$- \alpha N_I h + \beta Noh + LN_I \qquad (2)$$

2) Aphotic zone

$$\frac{\partial N'oh'}{\partial t} = \frac{\partial N'ou'h'}{\partial x} - \frac{\partial N'ov'h'}{\partial y} + \frac{\partial}{\partial x}\left(K'h'\frac{\partial N'o}{\partial x}\right) + \frac{\partial}{\partial y}\left(K'h'\frac{\partial N'o}{\partial y}\right)$$

$$+Kz\,(No - N'o) - \beta N'oh' + \gamma\,(No - N'o) + Bo \qquad (3)$$

$$\frac{\partial N'_I h'}{\partial t} = -\frac{\partial N'_I u'h'}{\partial x} - \frac{\partial N'_I v'h'}{\partial y} + \frac{\partial}{\partial x}\left(K'h'\frac{\partial N'_I}{\partial x}\right) + \frac{\partial}{\partial y}\left(K'h'\frac{\partial N'_I}{\partial y}\right)$$

$$+Kz\,(N_I - N'_I) - \beta N'oh' + B_I \qquad (4)$$

No : Organic nutrient concentration in euphotic zone

N_I : Inorganic nutrient concentration in euphotic zone

$N'o$: Organic nutrient concentration in aphotic zone

N'_I : Inorganic nutrient concentration in aphotic zone

K : Horizontal diffusion coefficient in euphotic zone

K' : Horizontal diffusion coefficient in aphotic zone

Kz : Exchange coefficient between surface layer and substratum

α : Rate of photosynthesis

β : Rate of decomposition

γ : Rate of precipitation

β' : Rate of decomposition in aphotic zone

Bo : Amount of organic nutrient salts leached per unit time and area

B_I : Amount of inorganic nutrient salts leached per unit time and area

LNo : Amount of organic nutrient salts received from land

LN_I : Amount of inorganic nutrient salts supplied from land

h, h' : Thickness of euphotic and aphotic zones

u, u': Flow rate in x-axis direction in euphotic and aphotic zones

v, v': Flow rate in y-axis direction in euphotic and aphotic zones

(B) COD Diffusion Equations

(1) Euphotic zone

$$\frac{\partial Sh}{\partial t} = - \frac{\partial Suh}{\partial x} - \frac{\partial Svh}{\partial y} + \frac{\partial}{\partial x}(Kh \frac{\partial S}{\partial x}) + \frac{\partial}{\partial y} (Kh \frac{\partial S}{\partial y}) - Kz (S - S')$$

$$+ a\alpha N_I h - K_1 Sh - K_3 S + L_S \qquad (5)$$

2) Aphotic zone

$$\frac{\partial S'h'}{\partial t} = - \frac{\partial S'u'h'}{\partial x} - \frac{\partial S'v'h'}{\partial y} + \frac{\partial}{\partial x} (K'h' \frac{\partial S'}{\partial y}) + \frac{\partial}{\partial y}(K'h' \frac{\partial S'}{\partial y}) + Kz (S - S') + K_3 S$$

$$-K'_1 Sh' - K_3 S' + S_B \qquad (6)$$

S : COD of euphotic zone

S' : COD of aphotic zone

a : Ratio of nutrient salt concentration and COD concentration in phytoplankton

K_1 : Reduction coefficient in euphotic zone

K_3 : COD precipitation rate in euphotic zone

K'_1 : Reduction coefficient in aphotic zone

K'_3 : COD precipitation rate in aphotic zone

S_B : Amount of COD leached per unit area and time

L_S : Amount of COD load received from land

(C) DO Diffusion Equations

(1) Euphotic zone

$$\frac{\partial Ch}{\partial t} = \frac{\partial Cuh}{\partial x} - \frac{\partial Cvh}{\partial y} + \frac{\partial}{\partial x}\left(Kh \frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial y}\left(Kh \frac{\partial C}{\partial y}\right) - Kz\,(C - C') + P\alpha\, N_I h$$

$$+ K_2\,(C_S - C)h - K_1\, Sh \qquad (7)$$

(2) Aphotic zone

$$\frac{\partial C'h'}{\partial t} = - \frac{\partial C'u'h'}{\partial x} - \frac{\partial C'v'h'}{\partial y} + \frac{\partial}{\partial x}\left(Kh' \frac{\partial C'}{\partial x}\right) + \frac{\partial}{\partial y}\left(Kh' \frac{\partial C'}{\partial y}\right) + Kz\,(C - C')$$

$$- K_1'\, S'h - D_B \qquad (8)$$

P : DO release rate by photosynthesis

K_2 : Reaeration coefficient

C_S : DO saturation rate

K_1 : Deoxidation coefficient

D_B : Amount of DO consumption per unit area and time

DETERMINING THE COEFFICIENTS

The coefficients used in the calculations were derived as follows:

Production

The production rate was calculated from the nutrient salt intake of the plankton obtained from an algal potential productivity (AGP) test performed on surface layer water collected at the site. The average value for the whole area was used.

Decomposition

The rate of decomposition was calculated from the difference between the initial concentration and the final concentration, after decomposition in a dark room, using samples collected at each water level (both surface layer and substratum). The average for the whole area was used.

Precipitation

The sediment sampler was deployed at the site from dawn to sunset. The precipitates were then collected and the concentration of specific substances in the precipitates determined. The rate of precipitation was calculated by comparing water qualities in the same water layer for both the surface layer and substratum. The average value for the whole area was used.

Leaching

Water just above the sea bottom, and cores of mud were collected at the site by divers. After the samples were brought back to the laboratory, the quantity leached was calculated from changes in the concentrations of substances in the collected water. The value was divided over three blocks for the substratum.

Oxygen Consumption

Samples were collected in the same way as for the leaching test. After the samples were returned to the laboratory they were reaerated. The amount of oxygen consumption was calculated from the changes in the concentrations of substances in the collected water.

Influx Load

The actual values were used as measured in 1976 in Chiba, Tokyo and Kanagawa Prefectures.

RESULTS OF DIFFUSION CALCULATIONS

Validation by Historical Records

Since the 1976 influx load values were used, a companion 1976 investigation of water quality (Figures 16-18) was used to check the validity of the model.

The coefficients used in the model were obtained from surveys performed in September, 1976 and August-September, 1977. These values are given in Table 2. The experimental values were corrected to those shown in the column on the right and are considered to more closely resemble current values.

The results from the model equations are shown in Figures 19-21. These results are outlined below.

COD

In the surface layer, the isopleths of 2 ppm and 3 ppm generally match current conditions, so the accuracy appears to be good. In the substratum, the values currently are 2 ppm or less near the center of the bay. The calculated values match those from the mouth to the middle of the bay. But the calculated values are 1-2 ppm higher than the current values in the innermost part of the bay.

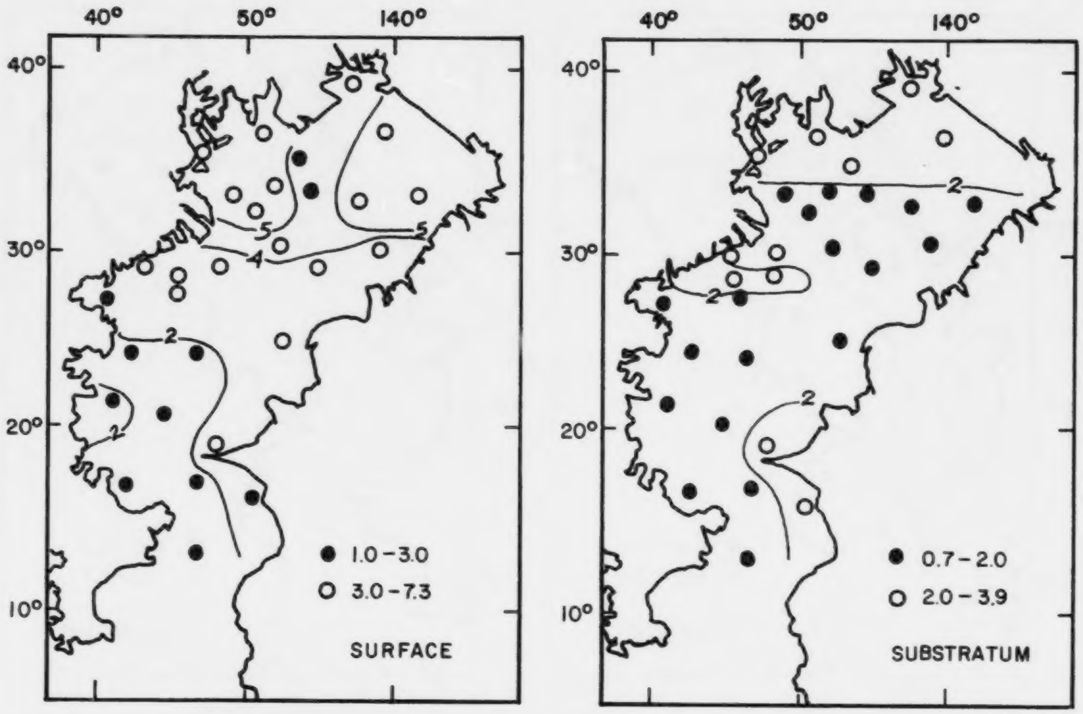

40°
50°
140°
40°
30°
20°
10°
5
5
4
2
2
● 1.0-3.0
○ 3.0-7.3
SURFACE
40°
50°
140°
40°
30°
20°
10°
2
2
2
● 0.7-2.0
○ 2.0-3.9
SUBSTRATUM

Figure 16. COD (ppm) average of July, August, September 1976.

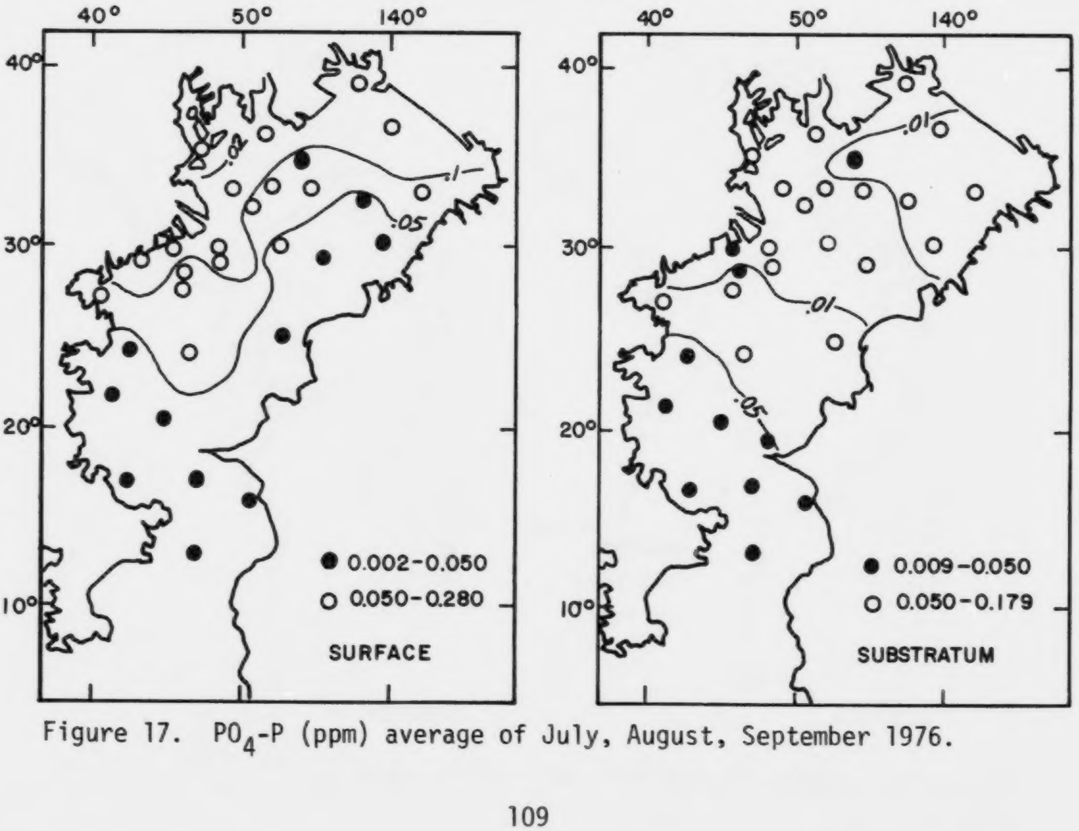

40°
50°
140°
40°
30°
20°
10°
.02
.1
.05
● 0.002-0.050
○ 0.050-0.280
SURFACE
40°
50°
140°
40°
30°
20°
10°
.01
.01
.05
● 0.009-0.050
○ 0.050-0.179
SUBSTRATUM

Figure 17. PO_4-P (ppm) average of July, August, September 1976.

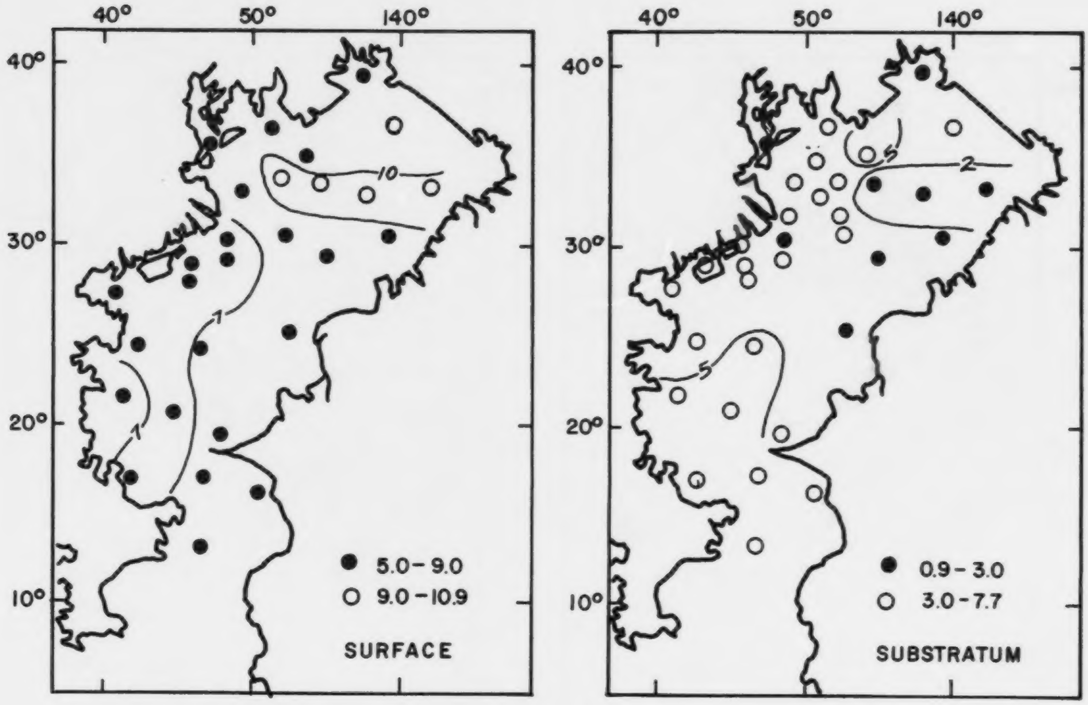

40°
50°
140°
40°
30°
20°
10°
10
7
7
7
● 5.0-9.0
○ 9.0-10.9
SURFACE
40°
50°
140°
40°
30°
20°
10°
5
2
5
● 0.9-3.0
○ 3.0-7.7
SUBSTRATUM

Figure 18. DO (ppm) average of July, August, September 1976.

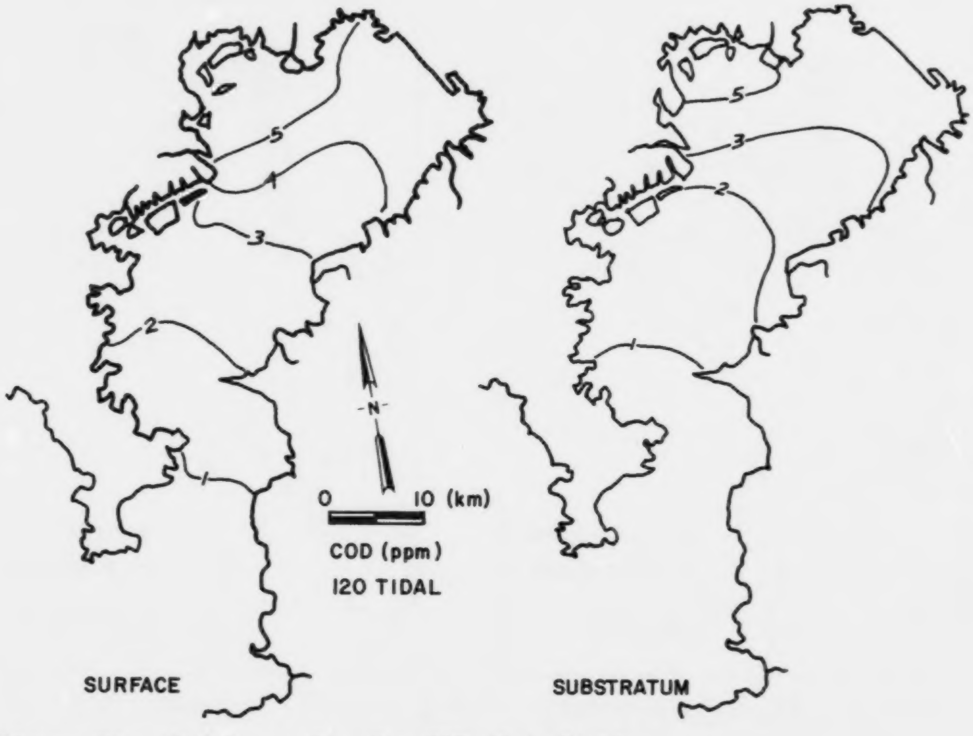

5
4
3
2
1
N
0
10 (km)
COD (ppm)
120 TIDAL
SURFACE
5
3
2
1
SUBSTRATUM

Figure 19. Model results for COD distribution.

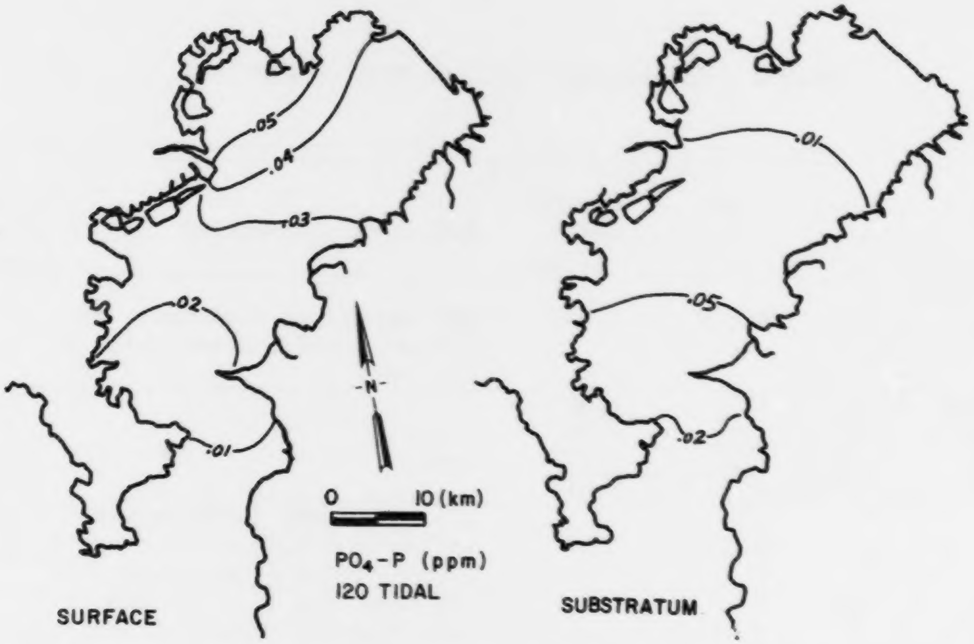

.05
.04
.03
.02
.01
.01
.05
.02
N
0 10 (km)
PO4-P (ppm)
120 TIDAL
SURFACE
SUBSTRATUM

Figure 20. Model results for PO_4-P distribution.

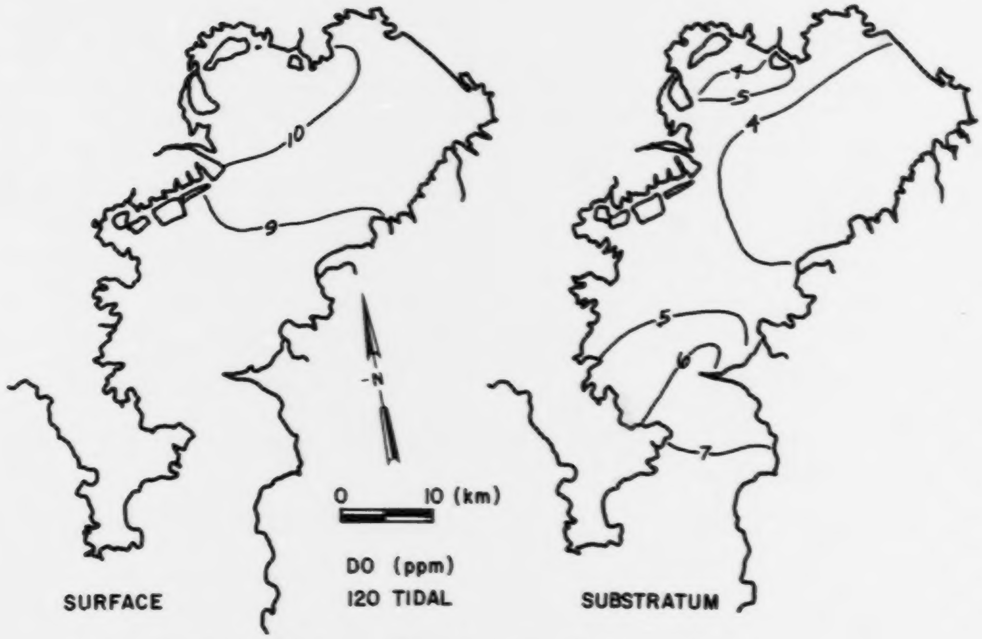

10
9
7
5
4
5
6
7
N
0 10 (km)
DO (ppm)
120 TIDAL
SURFACE
SUBSTRATUM

Figure 21. Model results for DO distribution.

TABLE 2. COEFFICIENTS USED AS MODEL PARAMETERS.

Coefficient	Block or layer	Item	Experimental value	Reason for Correction	Current value (used)
Production amount (/day)	Top	PO_4-P	0.409	30% reduction; remarkable increase in COD concentration in top layer	0.29
Decomposition amount (/day)	Top	COD	0.095	Same	0.095
		O-P	0.040		0.040
	Bottom	COD	0.078		0.078
		O-P	0.017		0.017
Precipitation amount (m/day)	Top	COD	0.57	30% reduction; large precipitation coefficient and drop in concentration of water quality.	0.40
		O-P	0.93	40% reduction (further reduction made from 30% since reproduction of surface layer phosphorus concentration still difficult).	0.56
	Bottom	COD	0.92	30% reduction; large precipitation coefficient and large reduction in concentration of water quality.	0.67
		O-P	1.10		0.77
Amount leached ($mg/m^2/day$)	Block 1	COD	650	(Maximum leached value of amounts 1976 and 1977 was 80% of the value statistically.)	650
		PO_4-P	65		65
	Block 2	COD	400		400
		PO_4-P	30	Same	30
	Block 3	COD	0	No distribution of sapropel	0
		PO_4-P	0		0
Amount of DO consumption ($gO_2/m^2/day$)	Block 1	DO	5.5	Same	5.5
	Block 2		4.0	(80% value according to calculation method as with the case of the amount leached).	4.0
	Block 3		0		0
	Total area	DO concentration	8.0 ppm	DO concentration corresponding to 100% DO saturation. Tokyo Bay, summer: 7.5-8.5 ppm)	8.0 ppm
Initial value (ppm)	Total area	COD	1.0	Monthly value for the bay from results of water quality measurements in 1976.	1.0
		PO_4-P	0.08		0.08
Remarks	Calculated tide: 120 tidal, diffusion coefficient: 10^5 cm^2/sec, dilution coefficient: 1.0				

PO_4-P

In the surface layer, the values resemble the current values of 0.02 ppm near the mouth of the bay. But, the current values from the middle to the innermost part of the bay are higher.

DO

In the surface layer, the calculated values for the whole area are more than 8 ppm. Actual measurements show supersaturated areas of 7 ppm or over, except in the western part of the bay. However, in the substratum, there are wide-ranging oxygen-poor conditions, especially in the area on the east coast of the inner bay where the calculated values were 4 ppm or less and their low values match the actual measurements. The 5 ppm isopleth at the mouth of the bay closely resembles the current value.

Summary

When the above results are summarized, it is evident that the model's accuracy was rather poor for COD in the substratum and for PO_4-P in the surface layer. But in general the calculated values matched the current values quite well from the mouth to the middle of the bay.

PREDICTION OF EFFECTS OF SAPROPEL REMOVAL

The following two cases were considered to determine the effects of removal of sapropel:

Case 1: Removal of 75 km^2 of sapropel for areas in the bay with particularly high leaching rates and oxygen consumption (Figure 22).

Case 2: Removal of 400 km^2 of sapropel from abiotic areas (Figure 22).

Figures 23 to 28 show the predicted changes in water quality for Cases 1 and 2.

In Case 1 the 4 ppm and 5 ppm predicted COD isopleths in the middle of the bay, when compared with current conditions, would shift inside the bay in the surface layer, as would the 3 ppm isopleth in the substratum. The prediction is, therefore, that Case 1 removal would purify the water in the innermost and middle parts of the bay. The PO_4-P would not change much in the surface layer - there would be an improvement of about 0.01 ppm in the inner bay. There would be few differences for DO in the surface layer, but in the substratum there would be fewer areas of 4 ppm or less and the oxygen concentration would increase near the mouth of the bay.

In Case 2, the COD concentration would decrease by 0.5-1.0 ppm from the middle to the innermost part of the bay, and there would be a remarkable improvement in water quality. The PO_4-P would show small values in the innermost part of the bay with large decreases in the substratum. There would be almost no change in the DO in the surface layer, but in the substratum there would be an increase in the oxygen concentration of about 1 ppm.

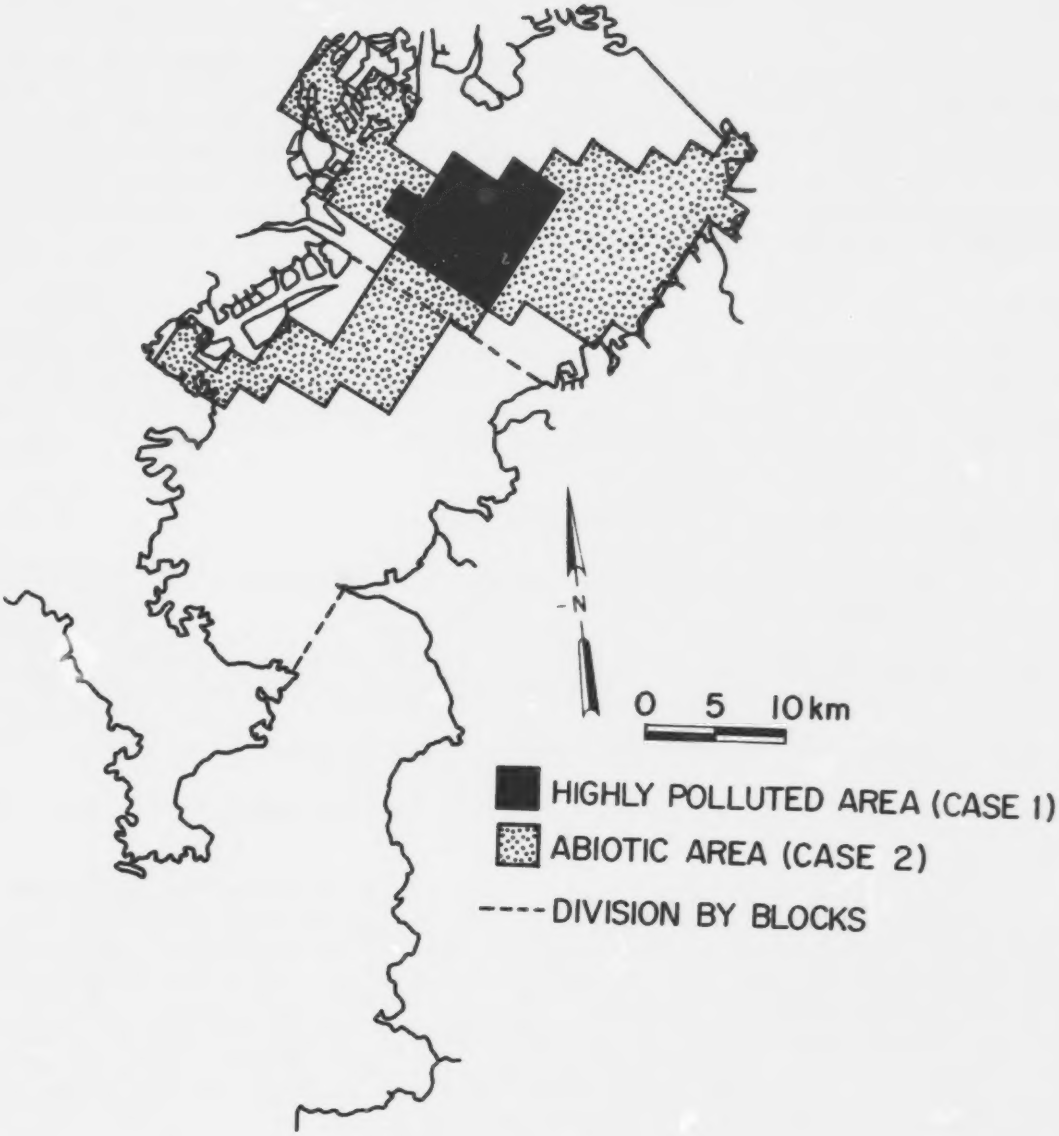
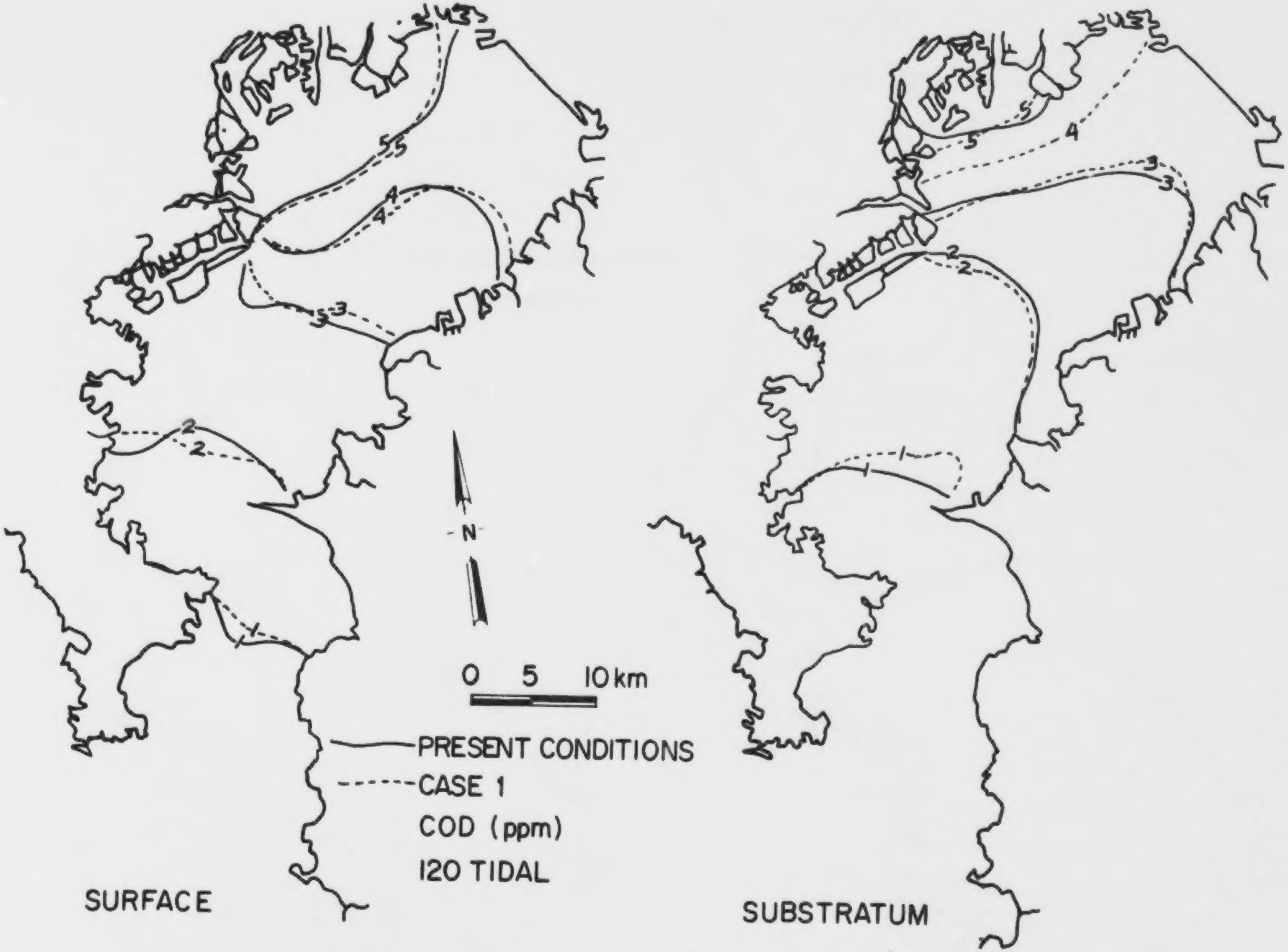

N
0 5 10km
HIGHLY POLLUTED AREA (CASE 1)
ABIOTIC AREA (CASE 2)
----DIVISION BY BLOCKS

Figure 22. Critical areas for sapropel removal.



PRESENT CONDITIONS
CASE 1
COD (ppm)
120 TIDAL
0 5 10 km
N
SURFACE
SUBSTRATUM
1
2
3
4
5

Figure 23. Predicted results of Case 1 (COD).

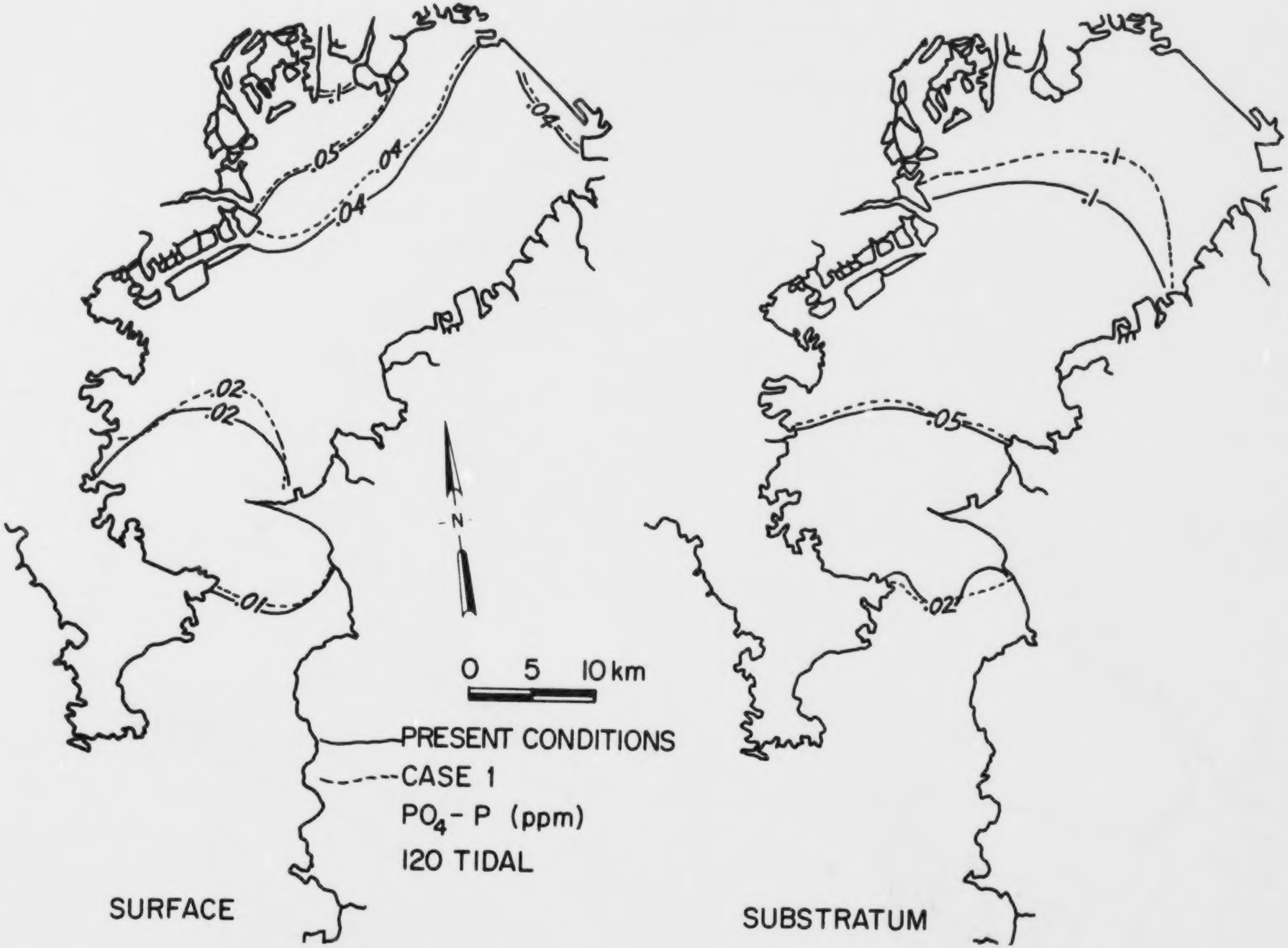

.1
.05
.04
.04
.04
.02
.02
.01
N
0 5 10 km
PRESENT CONDITIONS
CASE 1
PO4 - P (ppm)
120 TIDAL
SURFACE
.1
.1
.05
.02
SUBSTRATUM

Figure 24. Predicted results of Case 1 (PO_4-P).

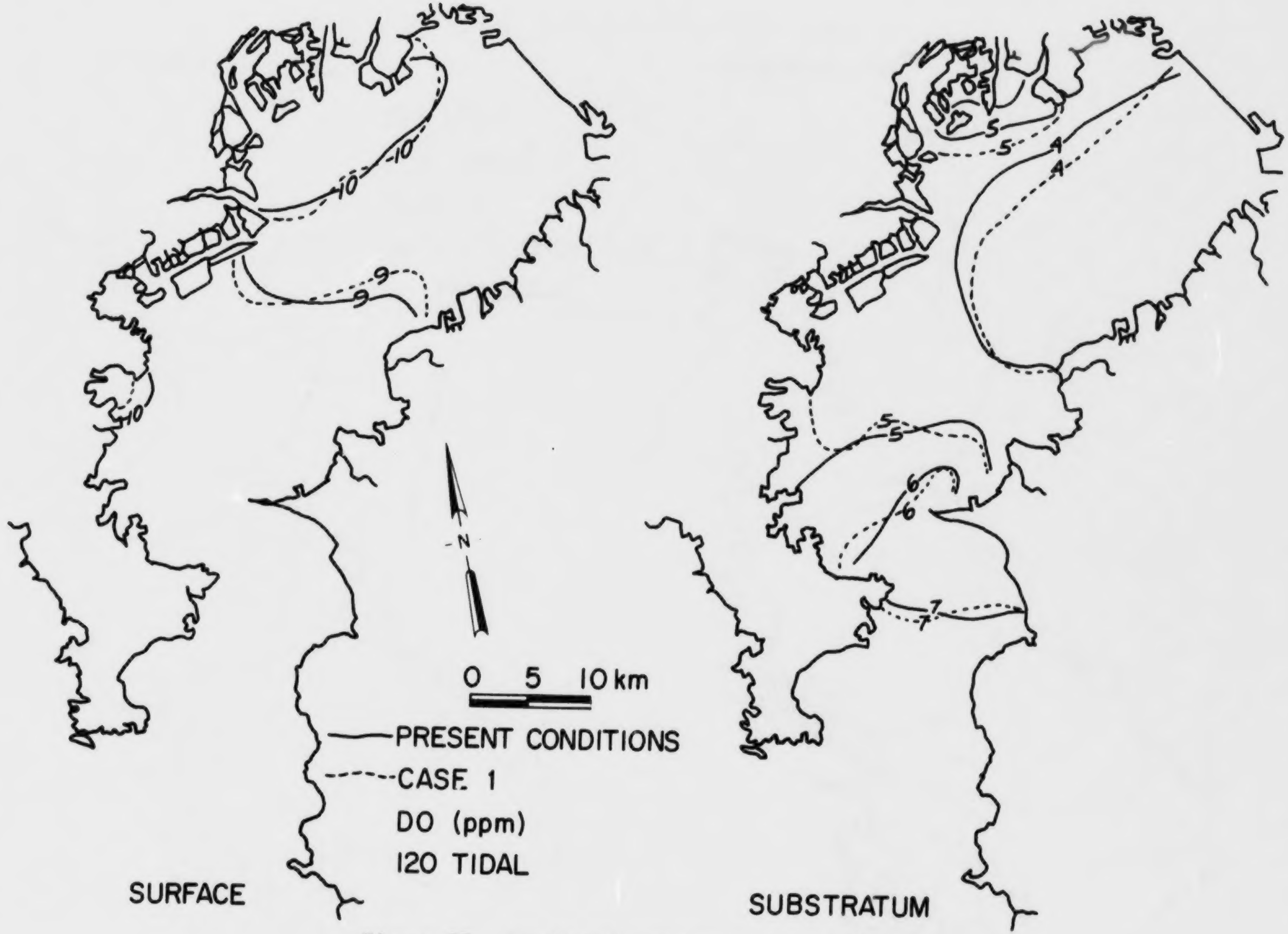

-10
-10
10
10
9
9
10
N
0 5 10 km
PRESENT CONDITIONS
CASE 1
DO (ppm)
120 TIDAL
SURFACE
5
5
5
4
4
5
5
6
6
7
7
SUBSTRATUM

Figure 25. Predicted results of Case 1 (DO).

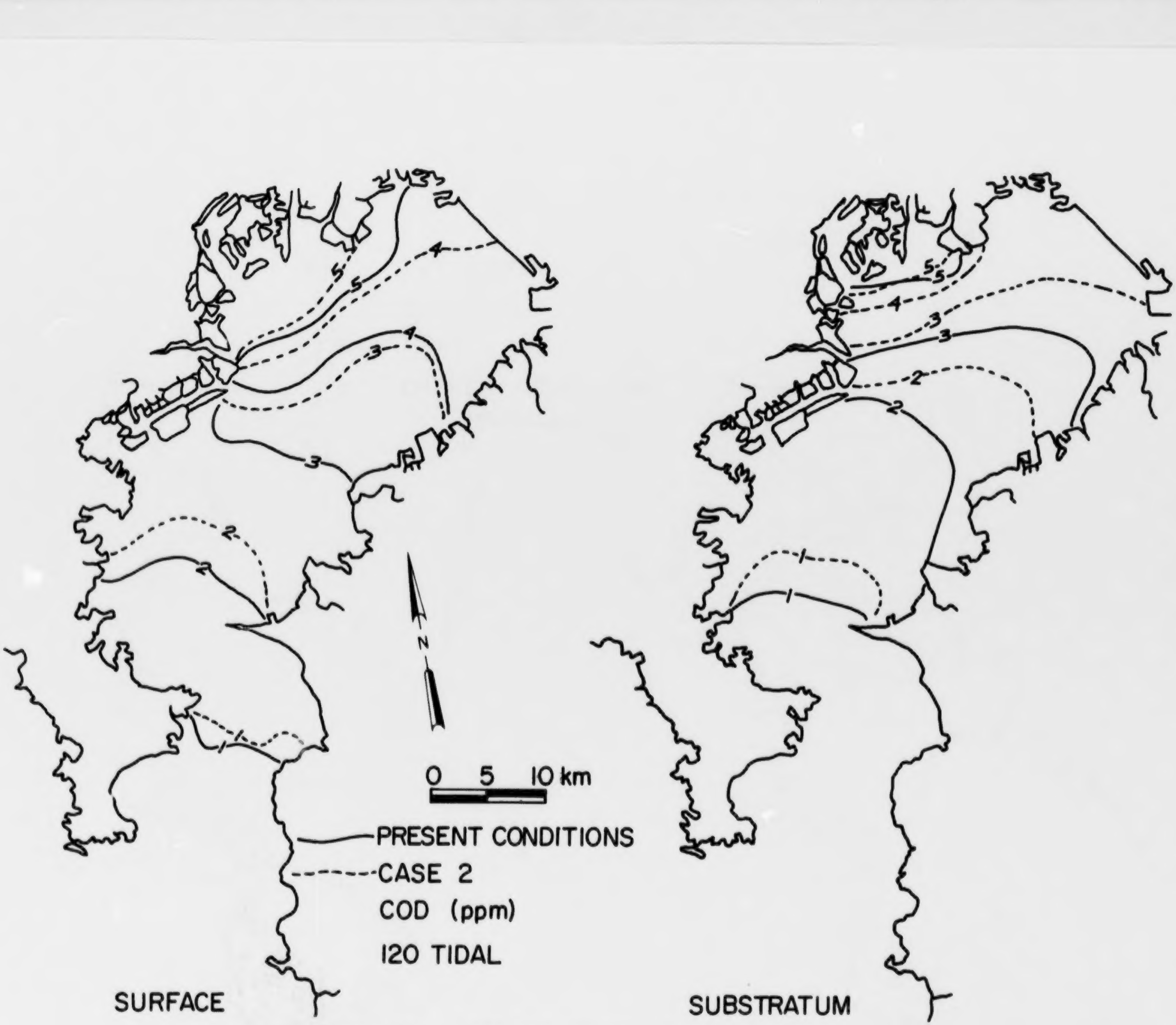

5
5
4
4
3
3
2
2
1
1
5
5
4
3
3
2
2
1
1
N
0 5 10 km
PRESENT CONDITIONS
CASE 2
COD (ppm)
120 TIDAL
SURFACE
SUBSTRATUM

Figure 26. Predicted results of Case 2 (COD).

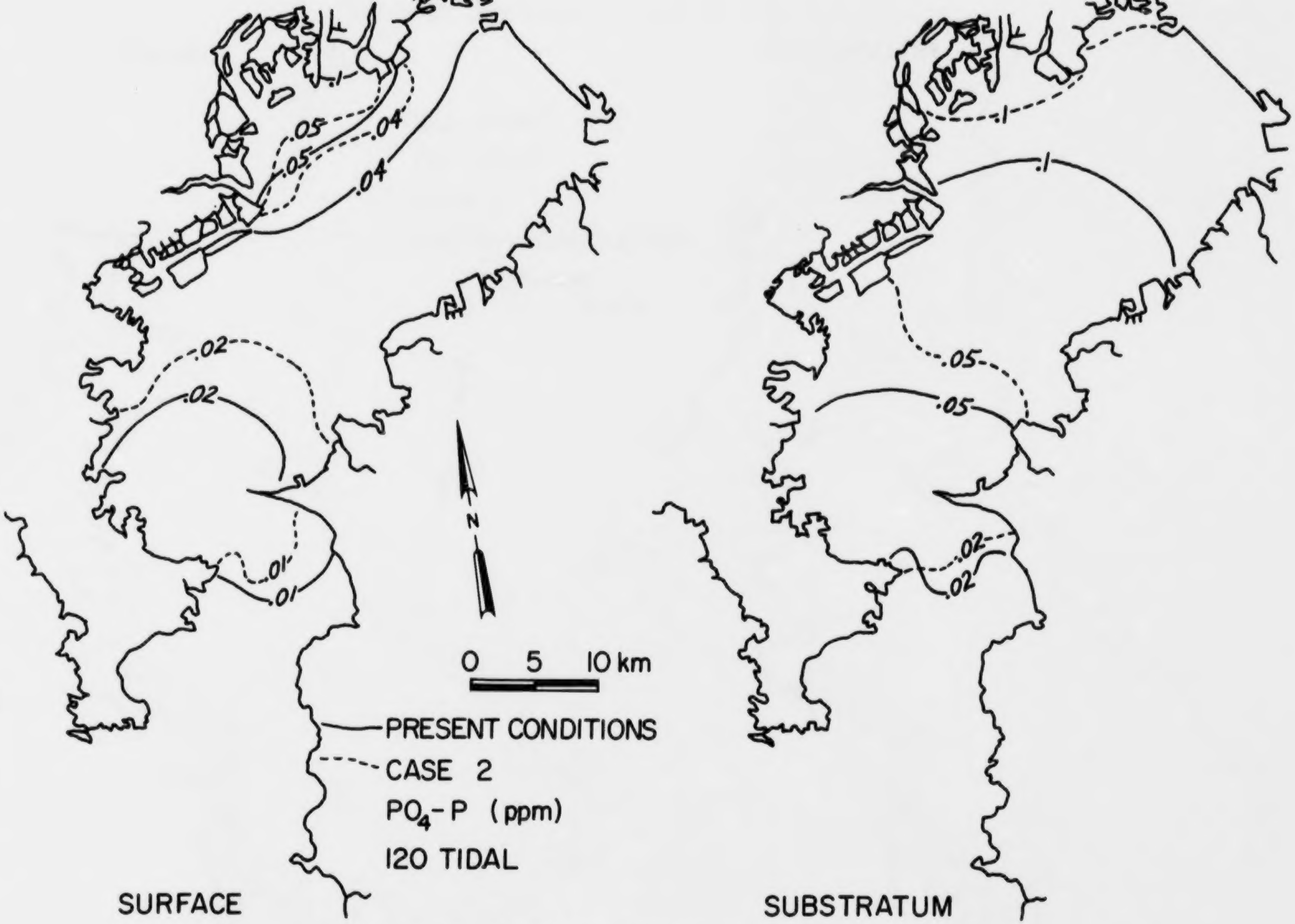

.1
.05
.05
.04
.04
.02
.02
.01
.01
.1
.1
.05
.05
.02
.02
N
0 5 10 km
PRESENT CONDITIONS
CASE 2
PO4-P (ppm)
120 TIDAL
SURFACE
SUBSTRATUM

Figure 27. Predicted results of Case 2 (PO_4-P).

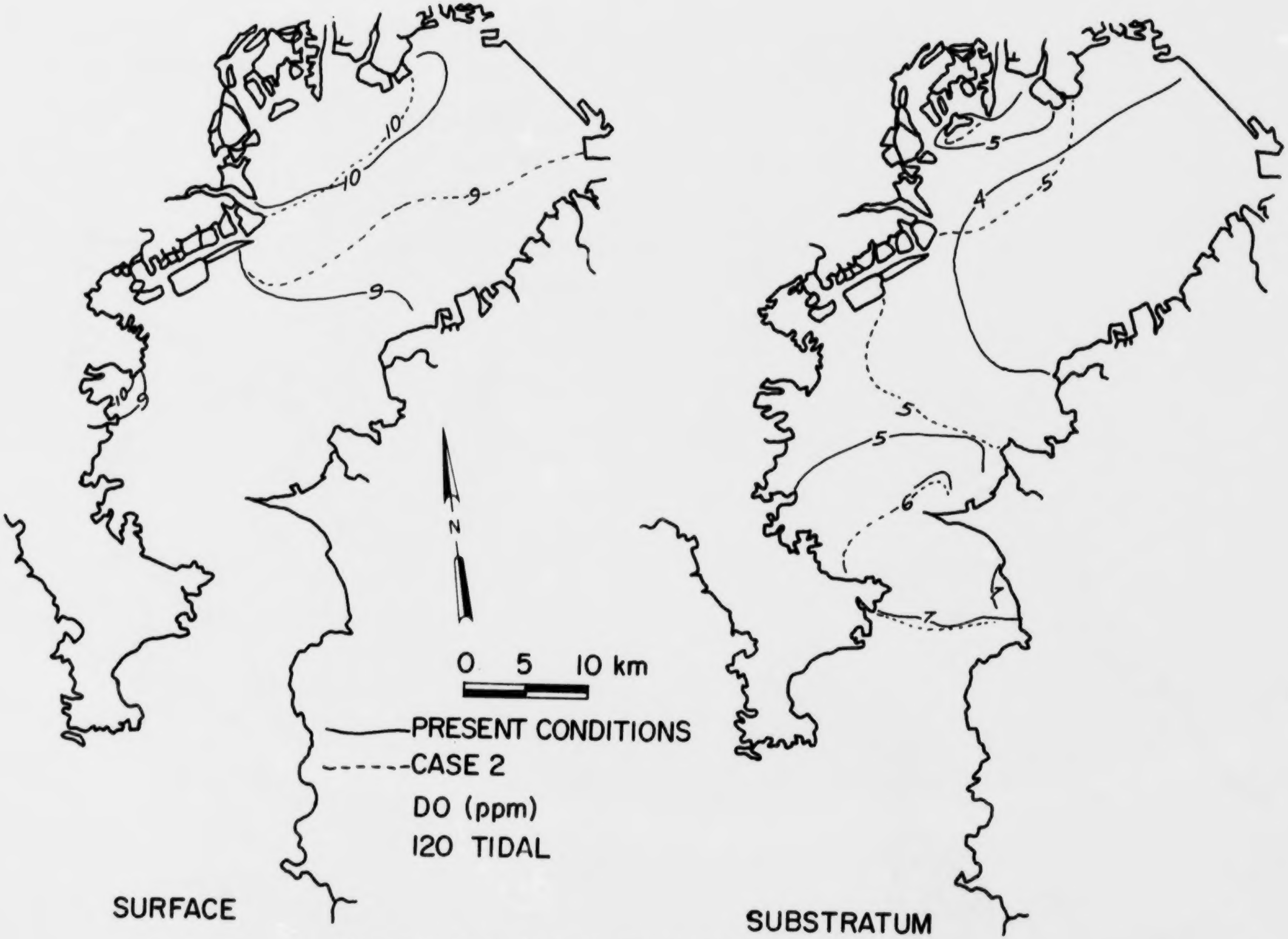

10
10
9
9
10
9
5
4
5
5
5
6
7
7
N
0 5 10 km
PRESENT CONDITIONS
CASE 2
DO (ppm)
120 TIDAL
SURFACE
SUBSTRATUM

Figure 28. Predicted results of Case 2 (DO).

Article 9 of the Basic Law for Environmental Pollution Control (1967) sets standards intended to develop and maintain water quality in public water areas with respect to the protection of human health and the environment. The health standards are uniform for all areas, but those concerning the environment include different standards for various types of water areas, such as rivers, lakes and the sea, depending on the purpose for which the water is used. By categorizing aquatic areas by use, the environmental quality standards for a critical area are clearly indicated. In Tokyo Bay, the water use categories for 18 regions are shown in Figure 29.

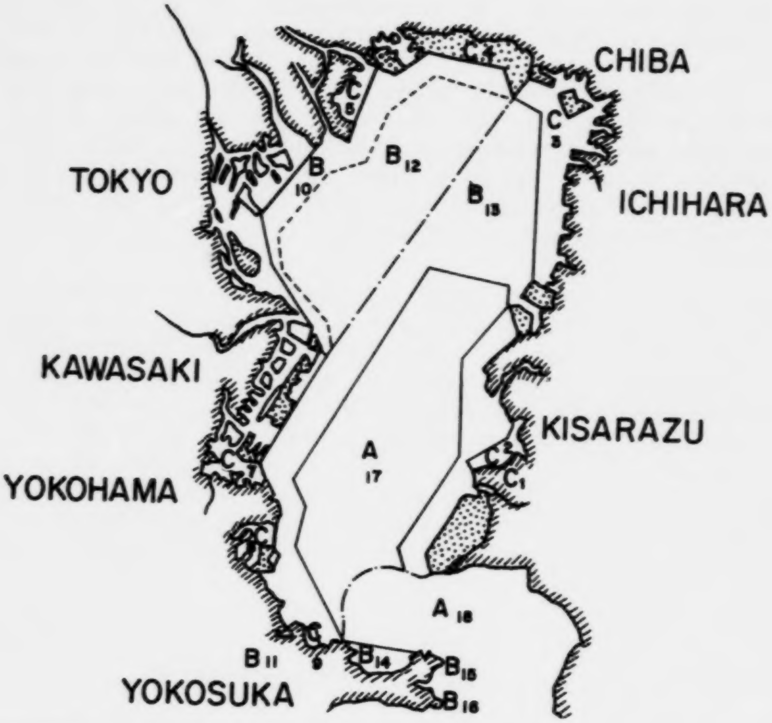
Table 3 shows the environmental quality standards for COD with respect to present conditions and the Case 1 and Case 2 predictions. The results indicate that, at present, the standards have been achieved for about 61% of Tokyo Bay as a whole. Among the various types of areas, the B type showed a particularly low achievement ratio of about 36%. For the simulated cases where the sapropel is removed, the improvement is naturally much greater when the removal area is larger. In Case 2 where 400 km^2 of sapropel is to be removed from the abiotic zones, it is difficult to achieve the required standards in more than 75% of the B category areas, but the percentage rises to 77% for the whole bay and such an improvement can be expected to create a fairly good aquatic environment.

TABLE 3. ENVIRONMENTAL QUALITY STANDARDS AND ACHIEVEMENT RATES (Unit: %).

Item	COD			
Use Category	A	B	C	Whole Bay
Standard values	2 ppm or less	3 ppm or less	8 ppm or less	
Current values	78.4	35.7	82.9	61.4
75 km^2 of sapropel removal (Case 1)	78.4	38.9	82.9	62.7
400 km^2 of sapropel removal (Case 2)	89.2	60.3	87.8	77.1

IMPLICATIONS OF MODEL RESULTS FOR THE BENTHOS

Figure 30 shows the relationship between the benthos and COD, sulfides, and ignition loss in bottom deposits. It is the same as Figure 12 but is marked with the predicted values from the simulation. The removal of sapropel improved ignition loss to 7%, sulfides to 0.2 mg/g and COD to 13 mg/g. It can be assumed that the number of organisms and the species diversity will therefore increase.


CHIBA
C4
C5
C3
TOKYO
B10
B12
B13
ICHIHARA
KAWASAKI
KISARAZU
C2
C1
A17
C7
YOKOHAMA
C
A18
C9
B11
B14
B15
B16
YOKOSUKA

		Utilization Category	Category
TOKYO BAY	1	Conservation of environment	C
	2	"	C
	3	"	C
	4	"	C
	5	"	C
	6	"	C
	7	"	C
	8	"	C
	9	"	C
	10	Fishery, class 2: industrial water and uses listed in C	B
	11	"	B
	12	"	B
	13	"	B
	14	"	B
	15	"	B
	16	"	B
	17	Fishery, class 1; bathing; conservation of natural environment and uses listed B, C	A
	18	"	A

Figure 29. Area ranked by environmental standards

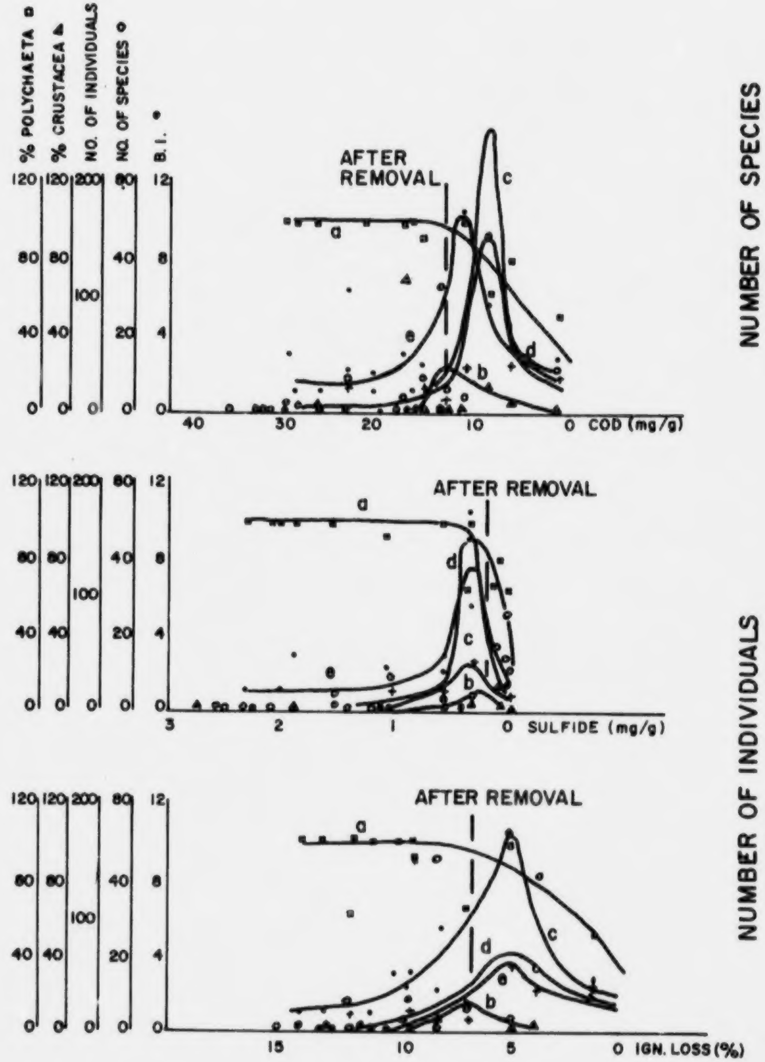


Figure 30. Correlations between zoobenthos and COD, sulfide and ignition loss, September 1976

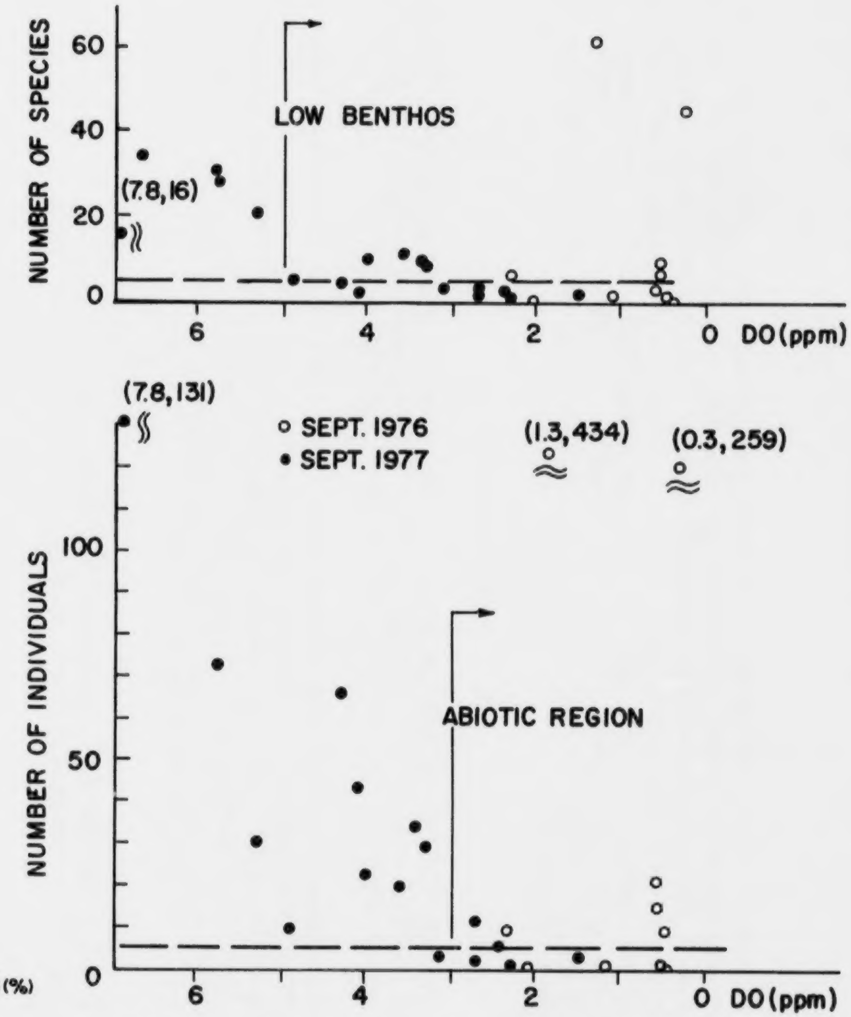


Figure 31. Relationship between benthos and DO in bottom water.

Figure 31 shows the relation between the benthos and water quality. At a DO concentration of 5 ppm, the number of species decreased to five or less and at about 3 ppm, the organisms numbered five or less. Since an increase in DO concentration in the substratum is one of the anticipated effects of the removal of sapropel, there should be a correspondingly better environment for organisms with improvement of bottom deposits and water quality. This will have a major effect on the recovery of marine life as outlined by the flow diagram in Figure 32.

CONCLUSION

Through studies of bottom deposits, water quality and biota in Tokyo Bay, it has become clear that sapropel, which has accumulated widely in the bay, has a major deleterious influence on the water quality and biota.

A mathematical eutrophication model was developed to investigate the effects of the removal of sapropel on water quality. The model was constructed based on data from experiments on leaching of organic detritus and nutrients from the deposits and DO consumption by the deposits. The results indicated that water quality would be greatly improved by the removal of sapropel, especially in the middle of the bay.

However, many questions still remain unanswered; among them are the long-term accumulation, decomposition rates and formation of new sapropel deposits after removal. This research will therefore be continued in the future.

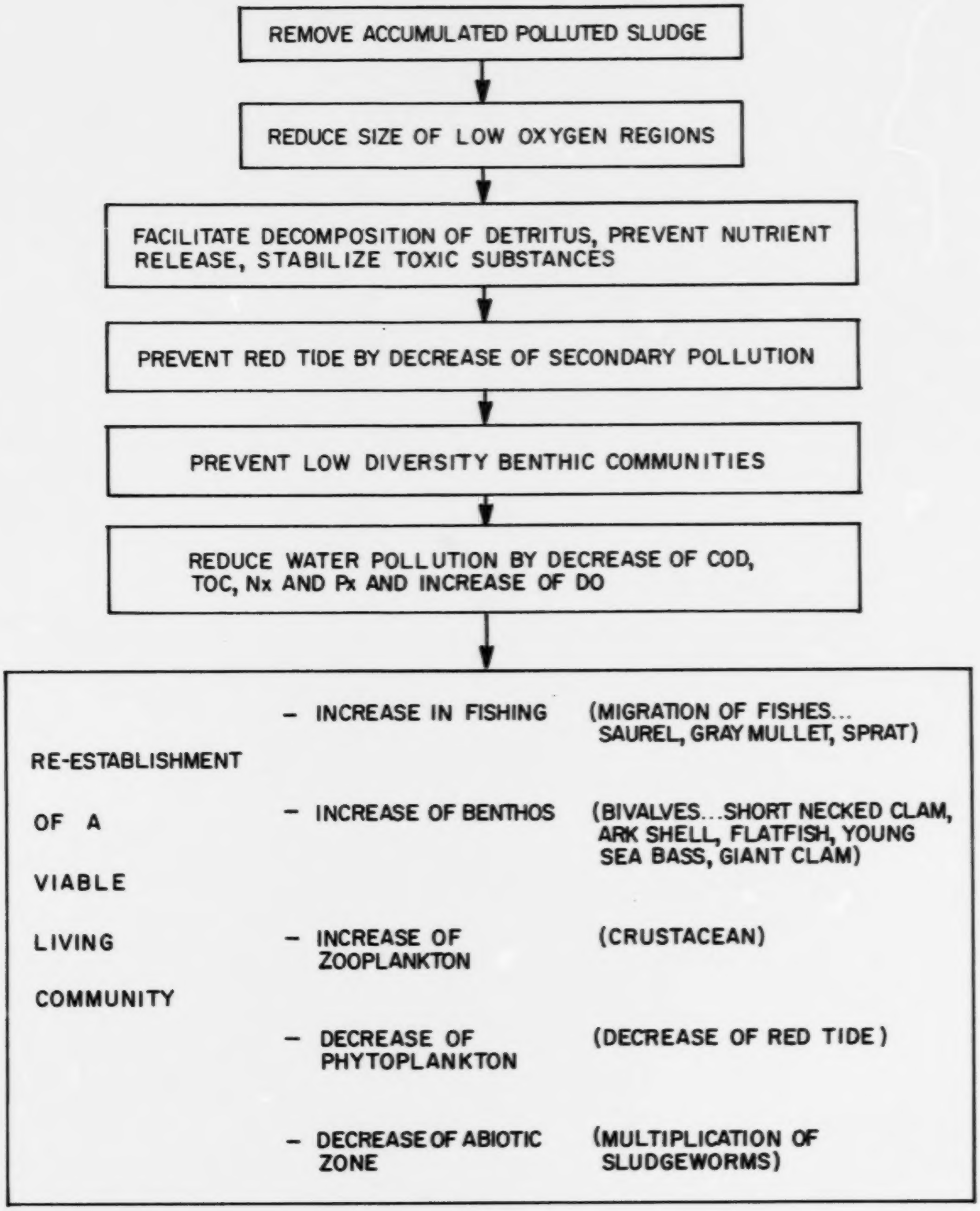

REMOVE ACCUMULATED POLLUTED SLUDGE
REDUCE SIZE OF LOW OXYGEN REGIONS
FACILITATE DECOMPOSITION OF DETRITUS, PREVENT NUTRIENT RELEASE, STABILIZE TOXIC SUBSTANCES
PREVENT RED TIDE BY DECREASE OF SECONDARY POLLUTION
PREVENT LOW DIVERSITY BENTHIC COMMUNITIES
REDUCE WATER POLLUTION BY DECREASE OF COD, TOC, Nx AND Px AND INCREASE OF DO
RE-ESTABLISHMENT OF A VIABLE LIVING COMMUNITY
- INCREASE IN FISHING
(MIGRATION OF FISHES... SAUREL, GRAY MULLET, SPRAT)
- INCREASE OF BENTHOS
(BIVALVES...SHORT NECKED CLAM, ARK SHELL, FLATFISH, YOUNG SEA BASS, GIANT CLAM)
- INCREASE OF ZOOPLANKTON
(CRUSTACEAN)
- DECREASE OF PHYTOPLANKTON
(DECREASE OF RED TIDE)
- DECREASE OF ABIOTIC ZONE
(MULTIPLICATION OF SLUDGEWORMS)

Figure 32. Flow chart showing effects of improved bottom deposits in Tokyo Bay.

RELEASE OF NUTRIENTS FROM LAKE SEDIMENTS

Ken Murakami, Chief
Kiyoshi Hasegawa, Research Engineer
Water Quality Section
Public Works Research Institute,
Ministry of Construction, Japan

ABSTRACT

Laboratory experiments were conducted on the release of nutrients from bottom sediments under various conditions. The relationships between nutrient release rates and nutrient content of sediments, temperature, redox potential of overlying water and other factors are discussed, based on the results of the experiments.

INTRODUCTION

In recent years, eutrophication has become a serious problem in many major lakes and reservoirs in Japan including Lake Kasumigaura, Lake Suwa and Lake Biwa. The phytoplankton blooms accompanying eutrophication not only obstruct direct water utilization, but scenic spots that used to have beautiful crystal clear water have lost their recreational value. The primary reason for eutrophication is the increase in nutrient loading from the drainage basin of each lake. But it is also well known that nutrient exchange between bottom sediments and overlying water is strongly related to increased eutrophication. Much research has been conducted on the role played by bottom sediments as a nutrient storehouse. However, the rates of nutrient transfer between the two phases have not been fully quantified. This makes it difficult to accurately estimate the nutrient budget of lakes, particularly for short time periods.

Laboratory experiments on the release of nutrients from bottom sediments used both disturbed samples and undisturbed core samples with flocculant sediments on the surface. The result of the experiments is discussed in this report.

EXPERIMENTAL METHOD

EXPERIMENTS USING DISTURBED SAMPLES

Both continuous and batch experiments were carried out using the apparatus shown in Figure 1. The vessel is 60 cm high and 40 cm inside diameter with a capacity of 75 liters. Tests were conducted by filling the bottom with a sediment layer approximately 10 cm high. In the batch apparatus a magnetic

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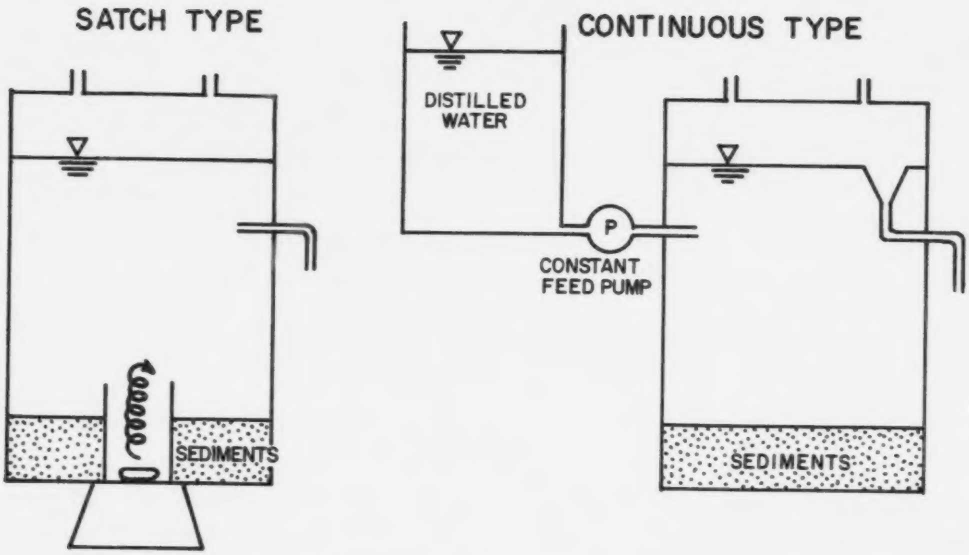

Figure 1. Apparatus for experiments with disturbed samples.

stirrer was used to examine the effect of the velocity of the overlying water on the release of nutrients. Distilled water was used as the overlying water in both batch and continuous systems.

In the batch test, the gas phase was released to the atmosphere in test runs under aerobic conditions. In test runs requiring anaerobic conditions, the water was purged by nitrogen gas and, at the same time, the gas phase (air) was replaced by nitrogen prior to the tests.

The continuous tests were performed to observe the variation in release rates when the overlying water of the bottom sediments changed from aerobic to anaerobic, and from anaerobic back to aerobic. To change the test water from aerobic to anaerobic, distilled water free from dissolved oxygen was supplied while, simultaneously the water in the vessel was purged with nitrogen.

The bottom sediment samples used in the tests were collected from Lake Kasumigaura, Lake Teganuma, Lake Toyanogata and the outer moat of the Imperial palace in Tokyo. Some sediment samples taken from the bottom of the Anakawa River and the Tsurumi River were also used.

EXPERIMENTS USING UNDISTURBED SAMPLES

Flucculant sediments on the lakebed surface could be related to the release of nutrients by bottom sediments. Since these flocculant sediments are difficult to collect by conventional methods, undisturbed samples were taken using core samplers, and the core tube itself was used as a test apparatus.

As shown in Figure 2, the sampler consisted of an acrylic plastic tube with an inside diameter of 5.2 cm and a length of about 66 cm. A sufficient number of samples were collected at a single station to provide 5 tubes for each test run. The samples were returned to the laboratory where the overlying water was carefully replaced by lake water filtered through a 0.45μ membrane filter.

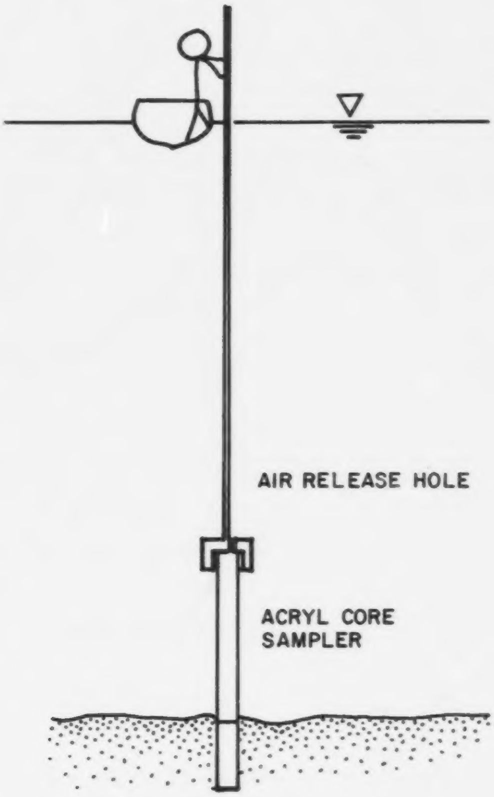

Figure 2. Core sampling for experiments requiring undisturbed samples.

In this experiment, since water volume in the core tubes was small, five tubes of samples were run simultaneously and used as replicates.

For the experiments under aerobic conditions, the overlying water was continuously aerated by a small amount of air during the test period. For the experiments under anaerobic conditions, two modifications were made. The first was to remove dissolved oxygen in the overlying water by using a nitrogen gas purge. The second was to add glucose to the water (20 mg/l at 20°C) after removing the dissolved oxygen. This helped maintain the anaerobic state.

The bottom sediments used in these tests were collected from Lake Kasumigaura. The sampling stations are shown in Figure 3. An attempt was made to maintain a consistent thickness of bottom sediments for each sample. But, in practice, they varied in thickness from 10 cm to 32 cm with an average of about 20 cm.

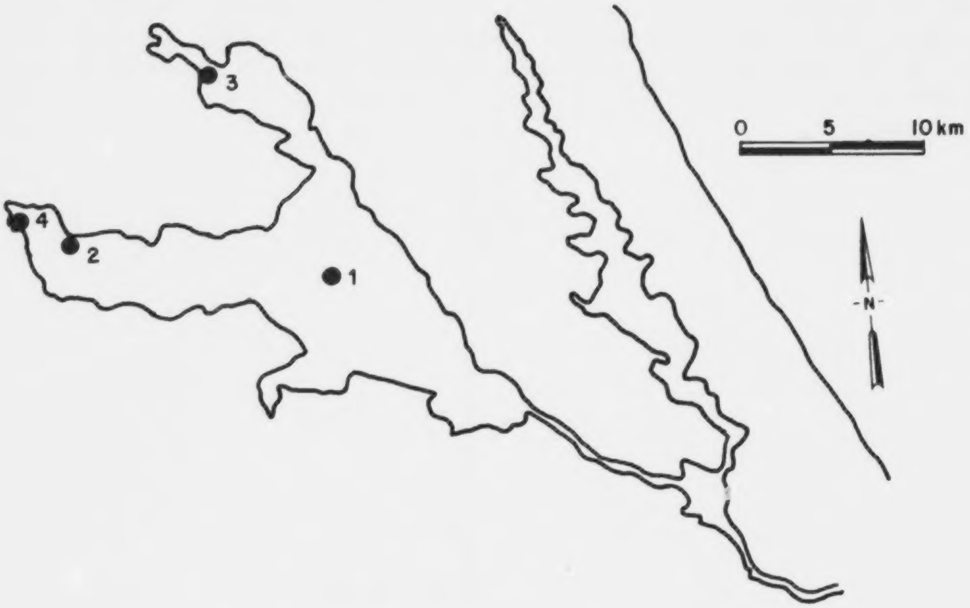

Figure 3. Sampling stations (core samples) in Lake Kasumigaura.

RESULTS AND DISCUSSIONS

PHOSPHORUS RELEASE

(1) The relationship between phosphorus release rate under anaerobic conditions and phosphorus content of the sediments.

The release of phosphorus from bottom sediments increases markedly when dissolved oxygen in water and the oxidation-reduction potential decreases. Figure 4 shows an example of results obtained from a continuous-type test with a disturbed sample. It is clear from the figure that the phosphorus concentration in the water begins to increase as the concentration of dissolved oxygen decreases. However, the release rate after the dissolved oxygen reaches zero is not constant. Figure 5 shows the time variation of the phosphorus release rate as calculated from the data shown in Figure 4. The general pattern is that, immediately after reaching an anaerobic condition, the release rate shows a marked increase and then decreases after about 6 days. The phosphorus release rate under anaerobic conditions is, therefore, time dependent. The average release rate during the first one- or two-week period after the dissolved oxygen declined to zero was compared to the phosphorus content of sediments, as shown in Figure 6. The release rate was in proportion to the phosphorus content of the sediments, and could be approximated by the following equation

$$Y = 10X \qquad (1)$$

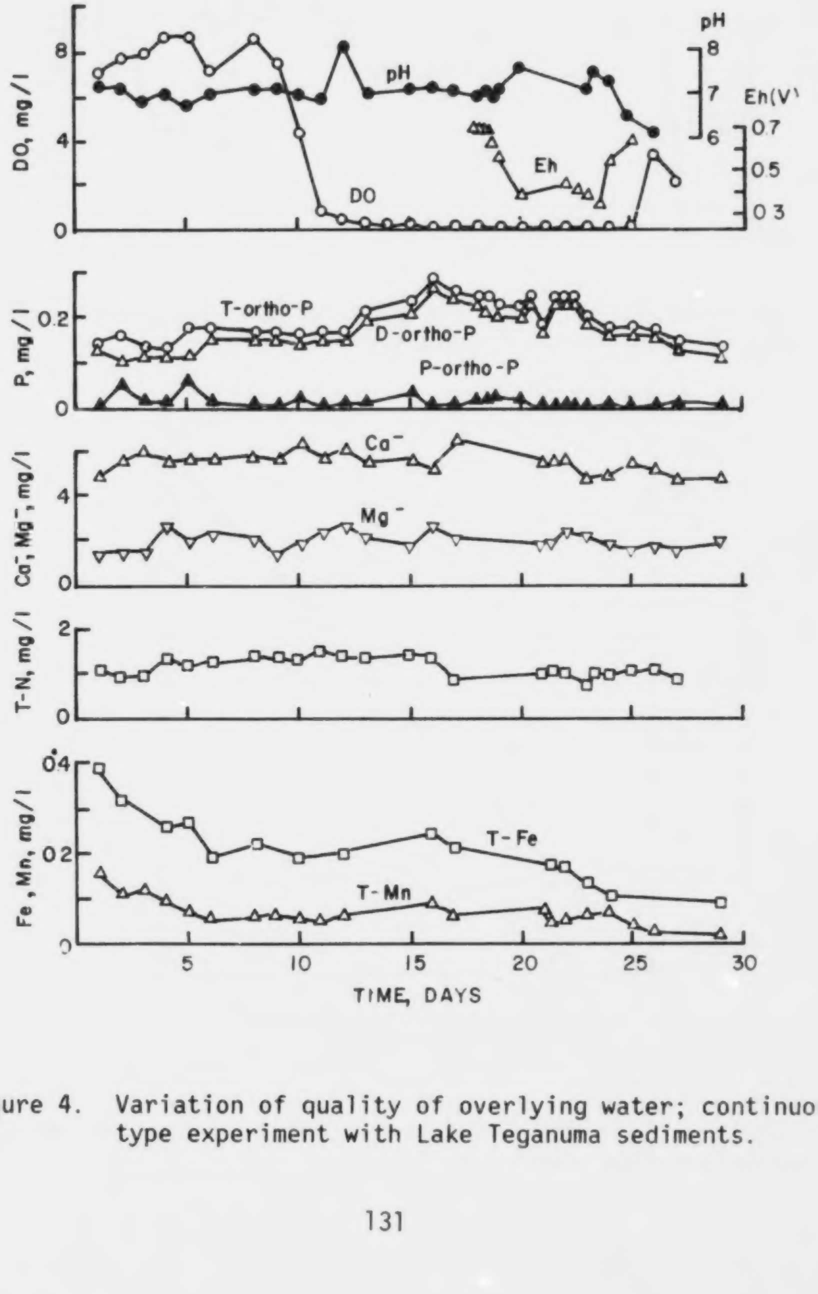

Figure 4. Variation of quality of overlying water; continuous type experiment with Lake Teganuma sediments.

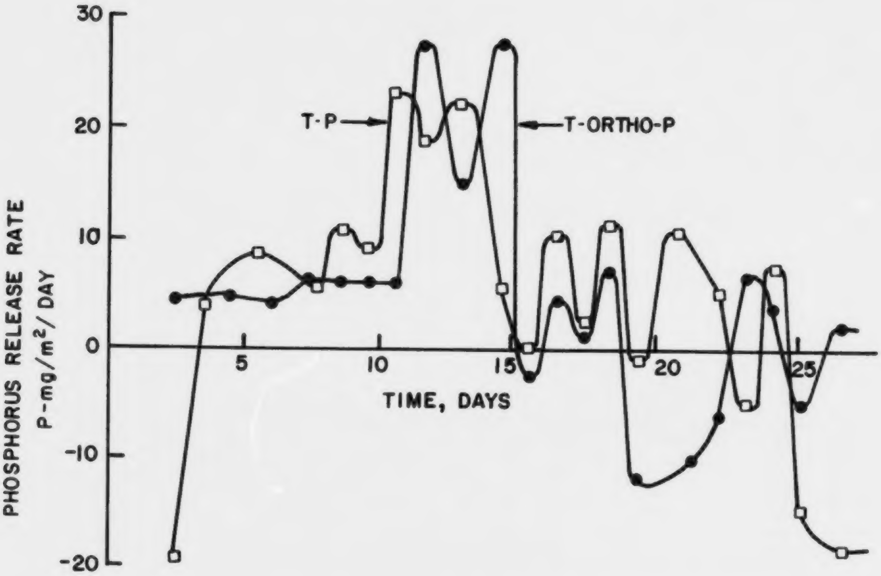

Figure 5. Time variation of phosphorus release rate; continuous type experiment with Lake Teganuma sediments.

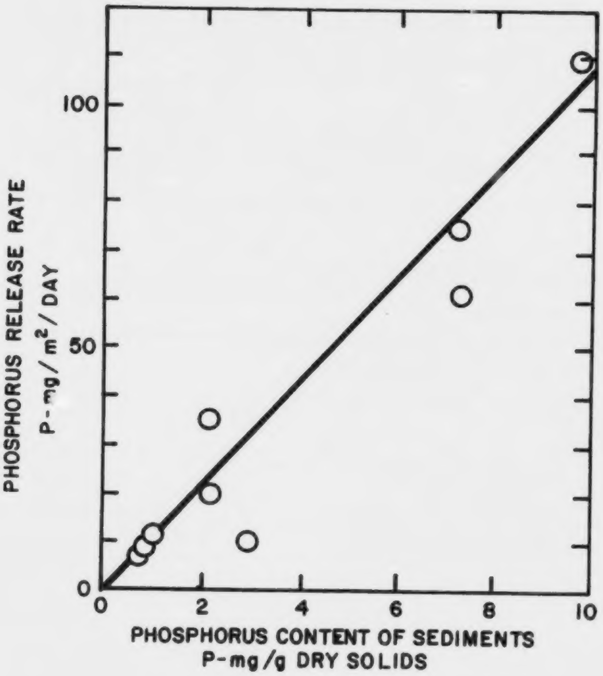

Figure 6. Relationship between phosphorus release rate under anaerobic conditions and phosphorus content of sediments; experiments with disturbed samples at 20°C.

where, Y is the average release rate of phosphorus (mg/m^2/day) under anerobic conditions at 20°C and X is the phosphorus content of the sediments (mg/g dry solids).

The same relationship was obtained by experiments using undisturbed samples with flocculant sediments. Figure 7 shows the relationship between the average release rates during the first 11 days after the start of the test at 20°C, and the phosphorus content of the surface sediments. It is interesting that equation (1) is also a good approximation for the flocculant sediments.

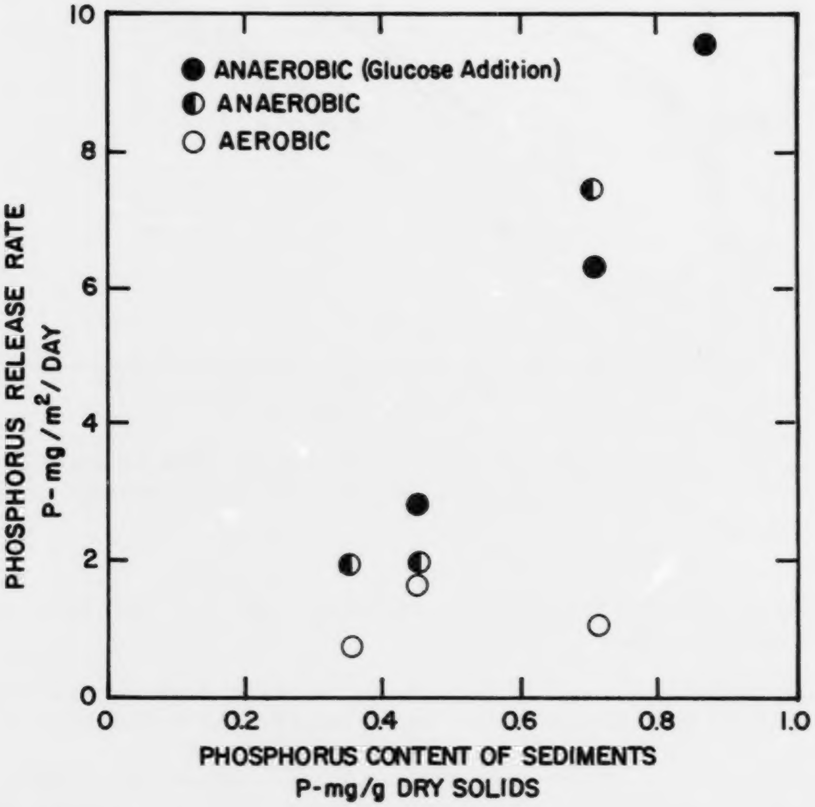

Figure 7. Relationship between phosphorus release rate and phosphorus content of sediments; experiments with undisturbed samples at 20°C.

The pattern of time variation of phosphorus release under anaerobic conditions, particularly the fast rate immediately after conditions become anaerobic, suggests that reduction of ferric phosphate is significantly involved in phosphorus release from the sediments. As shown in Figure 8, there is a good correlation between phosphorus and iron in the overlying water.

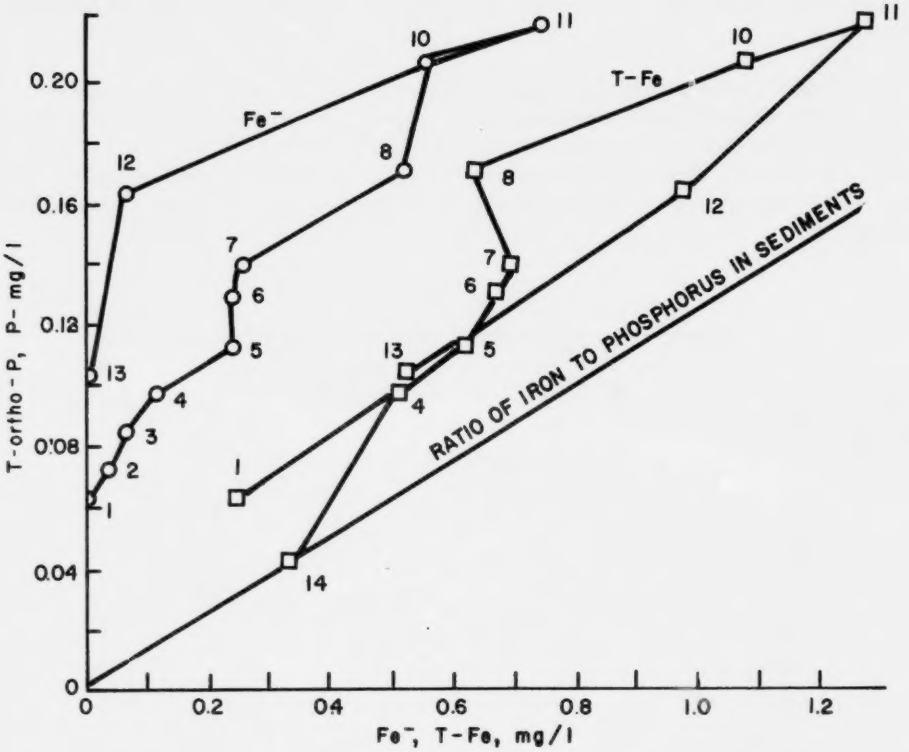

Figure 8. Relationship between phosphorus and iron in overlying water; continuous type experiments with Lake Teganuma sediments.

(2) The relationship between the release rate and the level of dissolved oxygen in overlying water.

As mentioned previously, phosphorus release from bottom sediments is a function of dissolved oxygen in the overlying water or a function of oxidation-reduction potential. Even though dissolved oxygen or oxidation-reduction potential is constant, the phosphorus release rate is time dependent. Therefore, it is difficult to find a strict relationship between the release rate and the dissolved oxygen level. A rough relationship was obtained based on a large number of data from the batch-type test using disturbed samples. Figure 9 is the result of this analysis, showing the relationship between release rate (at 20°C) and phosphorus content of the sediments for varying dissolved oxygen regimes in the water. The release rate may be expressed similarly to equation (1), i.e.

$$Y = aX \qquad (2)$$

where Y and X in equation (2) are the same as in equation (1). The value of coefficient "a" is roughly within the range shown in Table 1.

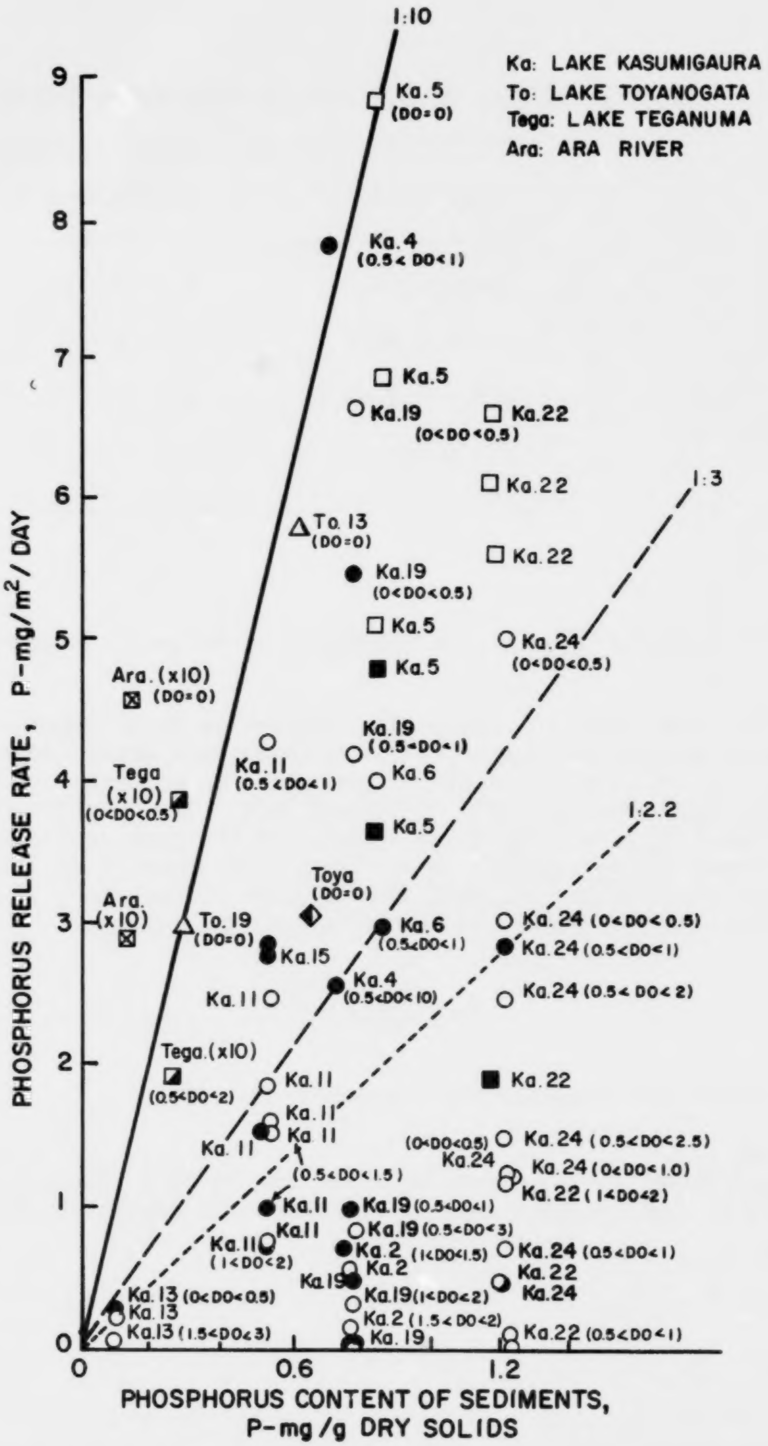

Figure 9. Effects of phosphorus content of sediments and DO level of overlying water on phosphorus releate rate; batch experiments at 20°C.

TABLE 1. EFFECT OF DO LEVEL ON PHOSPHORUS RELEASE RATE

Range of DO	Coefficient a
0 mg/L	$a \geq 10$
$0 \sim 0.5$ mg/L	$10 > a > 3$
$0.5 \sim 1.0$ mg/L	$3 > a > 2.2$
> 1.0 mg/L	$2.2 > a$

(3) The relationship between the release rate and water temperature

From the fact that the phosphorus content in water increases markedly in shallow lakes during the summer, it is assumed that water temperature greatly influences the release rate of phosphorus. Experiments using undisturbed samples were performed at 10°C, 20°C and 30°C. The results of plotting the average release rate for the first 11 days of the test against water temperature are shown in Figure 10. Under anaerobic conditions the release rate averaged 2.6 times higher at 30°C than at 20°C. The effect of water temperature on release rate is expressed in the following equation.

$$Y_T = Y_{20°C}\, \Theta^{T-20} \qquad (3)$$

where the coefficient Θ between 20°C and 30°C is 1.08.

Although fewer data are available for the 10-20°C temperature conditions, they imply $\Theta = 1.05$ for both aerobic and anaerobic conditions between 10°C and 20°C.

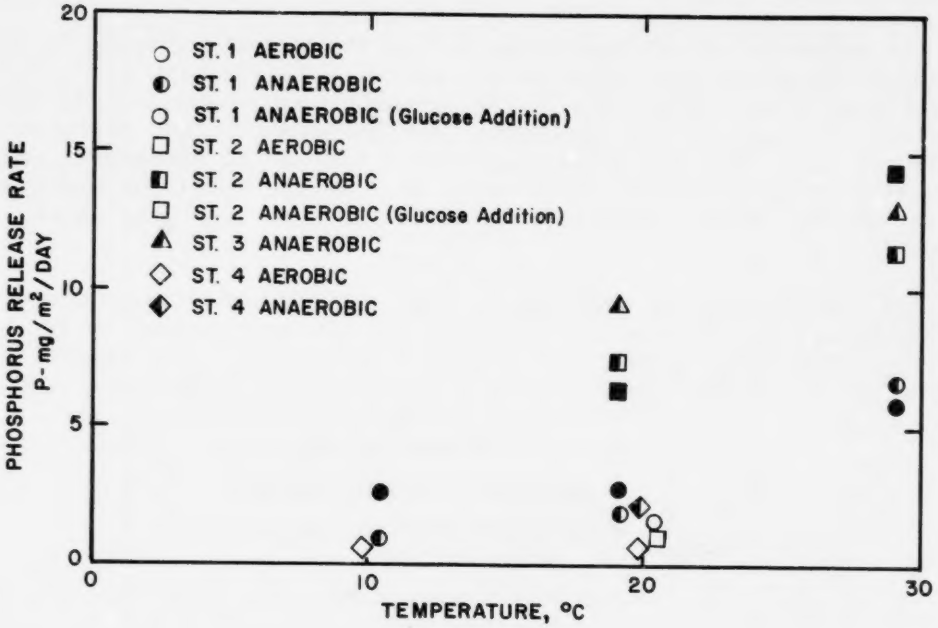

Figure 10. Effect of temperature on phosphorus release rate; experiments with undisturbed samples.

(4) Other factors influencing release rate

Other physical and chemical factors related to the release rate of phosphorus are the velocity of the overlying water and the diffusion coefficient in the sediments. When comparisions were made using the batch-type tests for release rates, with and without being subjected to water flow, there were no significant differences observed. This is because the majority of the tests were conducted under anaerobic conditions and the release of phosphorus was predominantly caused by the reduction of ferric phosphate rather than physical diffusion. The release under aerobic conditions and the steady release under long term continuing anaerobic conditions may be significantly influenced by the velocity of the water. No detailed studies were made on porosity or diffusion in the bottom sediments, but there is probably an influence similar to that of water velocity.

In addition, the release rate when flocculants were present on the surface of the sediment, showed virtually no differences compared to when flocculants were not present, based on data gathered under anaerobic conditions (Figures 6 and 7). Since few tests were run with flocculant sediments under aerobic conditions, no comparison of these results could be made with those obtained under anerobic conditions. Further studies on this subject will be made.

RELEASE OF NITROGEN

(1) The relationship between the release rate of nitrogen and the nitrogen content of the bottom sediments, and between the release rates and the oxidation-reduction potential of the overlying water.

The mechanism of nitrogen release from the bottom sediments is considered less time-dependent than that of phosphorus. Two factors are considered to control the release rate. One is the decomposition of particulate nitrogen to dissolved nitrogen in the sediments, and the other is the diffusion of dissolved nitrogen into the overlying water. Similar to phosphorus, major factors influencing breakdown of nitrogen will be the total nitrogen content of the sediments, aerobic-anaerobic condition of the overlying water, and temperature.

From the results of experiments made at 20°C water temperature, corresponding values for nitrogen release rate and nitrogen content of the sediments were found for both aerobic and anaerobic conditions. The results are shown in Figure 11. Under either aerobic or anaerobic conditions, the nitrogen

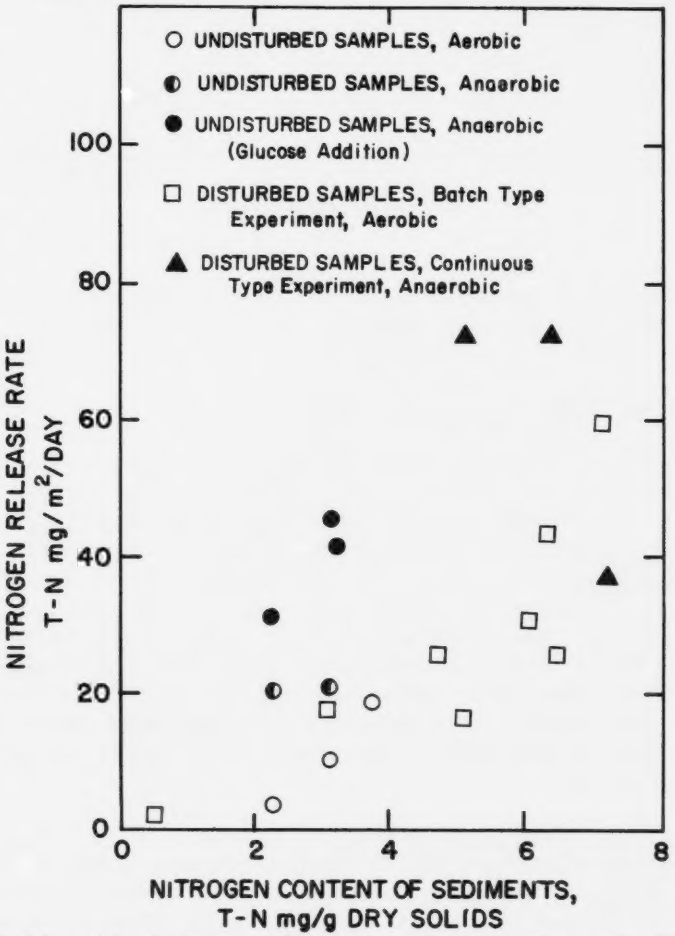

Figure 11. Relationship between nitrogen release rate and nitrogen content of sediments (20°C).

release rate is roughly proportional to the nitrogen content of the bottom sediments. However, under anaerobic conditions the nitrogen release rate was about 2 to 3 times greater than under aerobic conditions. The only possible reason for this is that the rate of decomposition of particulate nitrogen under anaerobic conditions is much higher than under aerobic conditions. This is an interesting result. The effect of flocculant sediment was not great since data from both disturbed and undisturbed samples show a similar trend.

(2) The relationship between nitrogen release rate and water temperature.

Similar to phosphorus, when plotting the average nitrogen release rate for the first 11 days of the water temperature experiments with undisturbed samples, the relationship shown in Figure 12 was obtained. The effect of water temperature on the nitrogen release rate was great. The release rates at 30°C under anaerobic conditions were about twice those at 20°C. Assuming the same relationship as equation (3) for the effect of temperature on the nitrogen release rate, the temperature coefficient θ was 1.07 between 20°C and 30°C under anaerobic conditions. Data for 10°C to 20°C under anaerobic and aerobic conditions were limited and therefore are less reliable, however, they were 1.06 and 1.095, respectively.

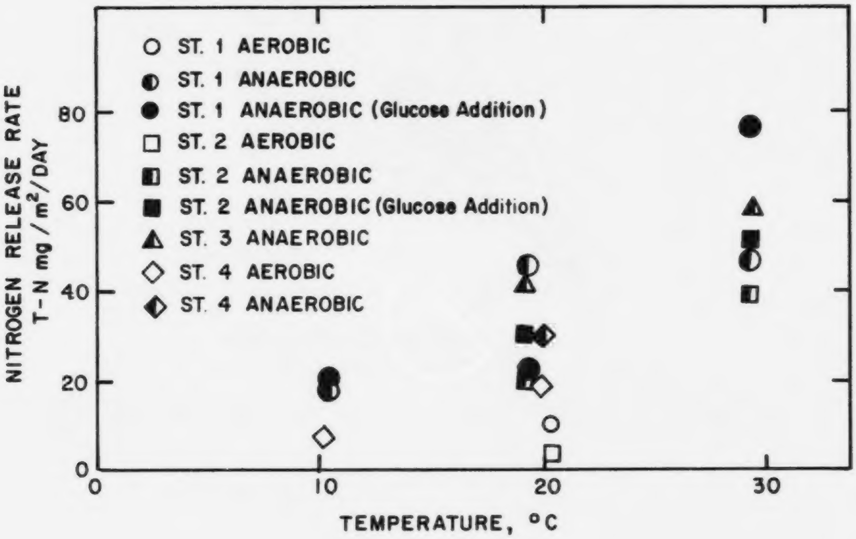

Figure 12. Effect of temperature on nitrogen release rate; experiments with undisturbed samples.

(3) The qualitative effect of water velocity on nitrogen release rate.

When comparing the nitrogen release rate with the phosphorus release rate, the nitrogen release was considered a more stable phenomenon, and thus more affected by diffusion. From the results of bottom tests employing disturbed samples, the qualitative effect of water flow on nitrogen release was examined. Comparison of the release rates with and without water flow over the sediment surface is shown in Figure 13. According to Figure 13, although the data show quite a spread, the nitrogen release under conditions of flow was larger than that without flow for about 80% of the data. Since rotation of the magnetic stirrer imparting velocity to the overlying water was arbitrarily controlled, it was impossible to express the effects of velocity quantitatively.

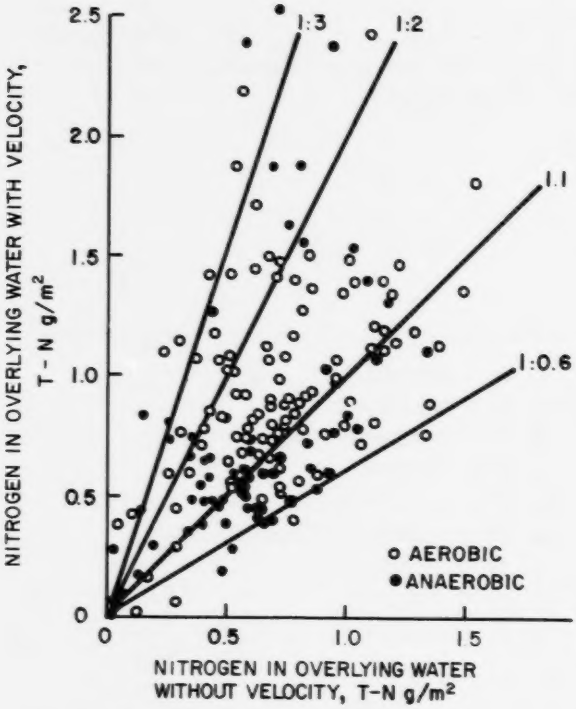

Figure 13. Effect of velocity on nitrogen release.

CONCLUSION

After careful examination of the test results for both phosphorus and nitrogen release from the bottom sediments of lakes, the following conclusions can be made:

(1) Phosphorus release

(a) The major factors that control the phosphorus release from sediment are the phosphorus content of the bottom sediments, the DO in the overlying water, and the temperature.

(b) Although the phosphorus release rate under anaerobic conditions is not constant, the average release rate during the first one or two weeks after the overlying water becomes anaerobic is in proportion to the phosphorus content of the bottom sediments. This relationship holds even when flocculant sediments are present on the surface.

(c) The phosphorus release rate is small under aerobic conditions in the water. When dissolved oxygen was over 1 mg/l, the rate dropped to about one-fourth of that under anaerobic conditions, based on results from experiments using disturbed samples.

(d) The phosphorus release rate is greatly influenced by temperature. Under anaerobic conditions, the release rate at 30°C is more than twice that at 20°C.

(2) Nitrogen release

(a) Similar to phosphorus, the nitrogen release rate depends on the nitrogen content of the bottom sediments, the DO of the overlying water, and the temperature. It is also affected to a certain extent by the velocity of the overlying water.

(b) Under both anaerobic and aerobic conditions, the nitrogen release rate is roughly proportional to the nitrogen content of the bottom sediments. But, the release rate under anaerobic conditions is two or three times higher than under aerobic conditions.

(c) The nitrogen release rate, similar to phosphorus, is also greatly influenced by temperature. The rate at 0°C under anaerobic conditions is about twice that at 20°C.

Some of the above conclusions are tentative. More studies are required to probe the release of nutrients from bottom sediments, especially under aerobic conditions, and to elucidate the influence of temperature.

ACKNOWLEDGEMENT

The experiments using undisturbed samples were performed by the Lake Kasumigaura District Office of the Ministry of Construction. Gratitude is expressed to those conducting the experiments.

TEST RESULTS FROM DEMONSTRATION DREDGING AND SPILLWATER TREATMENT AT HIRO HARBOR

Toshihiko Fukushima
Construction Bureau, Kure City Office

Tathuo Yoshida
Japan Bottom Sediments Management Association

INTRODUCTION

Hiro Harbor is located in the Hiro section of Kure-shi, a coastal city near Hiroshima. Formerly a small fishing village, Hiro became a major industrial import and shipping facility after Toyo Pulp Industries Ltd. built a large kraft paper mill on its waterfront. The wastewater from the mill polluted the area for a long time and, as a result, large quantities of polluted sediment now cover the harbor floor.

To deal with this pollution problem, the Kure-shi Municipal Office set up a pollution control committee for Hiro Harbor which has formulated a plan to clean up the area. This plan requires the dredging of 200,000 cubic meters of sediment, which will then be used as fill for a land reclamation project.

The implementation of this project requires the consent of the residents, particularly the fishermen, because of their concern over the environmental impact of the operation. The demonstration discussed here was designed to help obtain this consent and to determine probable environmental effects.

In early June of 1978, about 900 cubic meters of sediment were dredged for demonstration purposes. Ancillary tests on spillwater treatment, sludge solidification, and bioassay techniques were conducted. Results are reported in this paper.

OPERATIONAL METHOD

The dredging operation was performed as shown in Figure 1.* A special antipollution suction dredge dumped the dredged material into a barge. The barge was then anchored to the wharf near the experimental water treatment plant. The material in the barge was allowed to settle naturally over a two-day period. During this time the mixture separated into clean supernatant

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* Figures and tables for this paper are found at the end of the text, beginning on page 147.

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water and precipitated sludge. The water was pumped to the water treatment plant, and the sludge, after solidification, was hauled to a dump pit. It took only one hour to dredge the sediment, but about 10 days were required to treat the spillwater because of the small capacity of the test plant.

This public demonstration of sediment management was worthwhile because it increased the general public's understanding of the plan to clean up Hiro Harbor.

TURBIDITY

Turbidity generated by dredging is one of the most difficult problems to deal with. To control turbidity, several unique suction devices have been developed in Japan. In Hiro Harbor, one of these devices was used. It is similar to the one shown in Figure 2.

Two sets of augers with opposite pitches are mounted on a common axis within the suction device. The rotation of the screw forces the sediment to move to the center where it is picked up and pumped to the surface. When dredging with such a device, turbidity is kept low because of the smoothness of the cut and the near-perfect removal of loosened sediment particles. This is in contrast to conventional cutter-type devices.

Observations of turbidity were made during the Hiro Harbor demonstration. Measurements of SS concentration and the degree of turbidity were taken at 9 stations in the area, which was a square 100 m on a side beginning at the dredge site and running 100 m in the direction of diffusion. Samples were taken at three depths at each station. The results are summarized in Figure 4.

The maximum SS concentration near the dredge was 6 ppm. We estimate the increased concentration due to dredging as only 1-2 ppm by subtracting the background level of SS from the measured values. Therefore, it appears there is virtually no turbidity created by dredging with this equipment under the existing conditions. Similarly, the maximum degree of turbidity was only 0.6°.

The demonstration dredging statistics are:

swing width	50 m
swing speed	4 m/min
thickness of sediment	1.2 m
thickness of cut	0.4 m
amount of dredged sediment	130-180 m^3/hr
percent solids	20-40%
dredged water	650 m^3/hr

The sediment thickness and pulpy quality resulted in easy dredging.

SEDIMENT PROPERTIES

Four samples were examined to determine the particulate composition of the sediment.

Locations of sampling stations

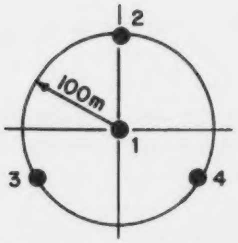


Table 1 shows that the grain size of the sediment at Hiro Harbor is very small—over 60% of particles are under 6.8 μ. This is smaller than results suggested by laboratory tests last year which gave a D_{60} of 12 μ and a D_{30} of 2.4 μ.

Physical and chemical properties of the sediments obtained last year are given in Table 2. The level of toxic substances is negligible except that comparatively high contents of Pb (20 ppm), Zn (349 ppm) and As (5.81 ppm) should be watched.

Organic matter content is fairly high as indicated by the high COD content—the highest we have ever seen (Table 3). This may be due to the pulpy nature of the sediment. The content of the n-Hexane extractions is also high. This means that the sediment contains large quantities of organically derived oils. The ammonia-nitrogen content is unexpectedly small. This may be due to conversion of sedimented nitrogen components to easily dissipated ammonium. The high hydrogen sulfide content is responsible for the bad odor of the dredge water(Figure 5).

The physical properties of the dredge water were analyzed and results are given in Table 4.

Data in Table 4 and Figure 6 were obtained from laboratory tests of dredge slurry containing 10, 20, 30 and 40% solids. These data show the relationship between SS and various chemical and physical variables. Use of these data with field measured SS concentrations provides an estimate of the other variables. It is extremely difficult to measure these variables directly.

WATER TREATMENT

A schematic of the water treatment plant is given in Figure 7. It is a demonstration plant which was set up to treat the supernatant water from the spoils barge after the sediment had settled naturally in the barge hold (Figure 8).

It took 45 hours to obtain a 1.8 m depth of clean supernatant water. From this, the mean settling velocity can be calculated as follows:

$$W = 1.8 \div 45 = 0.04 \text{ m/hr} = 0.0011 \text{ cm/sec} \quad (1)$$

The plant was operated continuously for 8 hours a day. A circulating high-speed vertical type filter was used. Sodium hypochlorite, a food additive, was used for chlorination. A commercial product (Kurifloc PA-331) was used as a polymeric coagulant. Table 5 shows the additives and operating parameters for the system.

A precipitation test was carried out using a 1000 cc cylinder. Results with and without a coagulant added are shown in Figure 9. The mean settling velocity without coagulation was

$$W = (165 \text{ cm} - 140 \text{ cm}) \div 6 \text{ hr} = 0.042 \text{ m/hr} = 0.0011 \text{ cm/sec}, \qquad (2)$$

which agrees with the first value derived from measurements in the barge. These slow settling rates will cause difficulty in handling the Hiro Harbor sediments. The SS concentration in the supernatant water was 30-60 ppm by weight.

Table 6 shows the water quality at each stage of operation. The initial SS concentration was reduced to 8-12 ppm after coagulation, and to 1-5 ppm after filtration. From the data presented in the table, it can be seen that COD_{Mn} concentration and the total phosphorus concentration were reduced in proportion to the SS removed. This is illustrated in Figure 10.

Since the guidelines for discharging this kind of water are generally 30 ppm SS and 20 ppm COD_{Mn} the water remaining after coagulation of the sediment might have been clean enough to discharge. However the T-N concentration was too high. Its value did not drop at all until the filtration stage, although the SS concentration was already reduced. Only in the last stage, when subjected to chlorination with sodium hypochlorite, did it drop adequately (Figure 11).

The results of the chlorination test are shown in Figure 12. The concentration of NH_4-N decreased rapidly as the active concentration of chloride ion exceeded 150 ppm, reaching a low point near 200-230 ppm of chloride ion. At the zero level the minimum residual chloride became noticeable.

In the experimental operation, the effect of activated carbon powder on the reduction of SS and COD_{Mn} at each stage was also examined. The results are summarized in Table 7. It was found that activated carbon powder has no effect on the efficacy of the other processes.

From the above results, it is concluded that the greatest problem in supernatant water treatment is the exceedingly high content of NH_4-N in the dredge water. It appears that this concentration derives from bacteria which convert the organic matter in the sediment into ammonium ion, and the ammonium ion then dissolves in the interstitial water of the sediment. Since concentrated ammonium ion is toxic to fish, it is necessary to reduce it below toxic levels. For this purpose chlorination is required. Usually the guideline for water quality when discharging dredge water is concerned with two items—SS concentration and COD. But in the case of Hiro Harbor, NH_4 must be added to the guideline.

SOLIDIFICATION OF SLUDGE

The sludge, which settled on the bottom of the barge, was not discharged in an untreated state. It was solidified at the site and then hauled by truck to a dump site. The commercial compound "Chemikolime 235" was used as a solidifier. This was combined with the sludge as follows:

	C-235	Water	C/Water	Concentration
Batch:	500 kg	400 kg	80%	0.89 t/m^3

Since 85 m^3 of solidifier were used to treat 490 m^3 of sludge, the mix percentage was 16.5%. The solidifier was mixed into the sludge with a clamshell bucket. The compression strengths of the solidified sludge are plotted in Figure 13. The deeper layer is stronger than the shallow one. This may be due to precipitation of the solidifier. Good agreement between the laboratory test data and on-site tests are shown by Figure 14. The good results may be attributed to selection of the proper solidifier and thorough mixing with the clamshell.

CONCLUSIONS

The results of experiments at the dredging site and water treatment at Hiro Harbor led to the following conclusions. Turbidity from dredge spoils, which is the most troublesome aspect of dredging, can be negligible if an anti-pollution dredge with a special suction device is used. The most important aspect of water treatment in this area is the exceedingly high ammonium ion content in the supernatant dredge water. To reduce it to permissible levels chlorination is the best treatment but, carbon is necessary to absorb the residual chlorine.

This demonstration was valuable in increasing public awareness of environmental problems and providing a basis for understanding the plan to clean up Hiro Harbor.

TABLE 1. GRAIN SIZE OF SEDIMENT

	Station 1	Station 2	Station 3	Station 4	Mean
Maximum Size μ	2	0.84	0.42	0.84	1.03
D_{60}	8.4	6.9	6.7	5.3	6.8
D_{30}	3.6	4.0	3.2	2.1	3.2

TABLE 2. PHYSICAL AND CHEMICAL PROPERTIES OF THE SEDIMENT

Soil property	Color		-	Black and Sticky
	Odor		-	Rotten
	pH	H_2O	-	7.80
		KCl	-	7.58
	Water Content Ratio		%	436.8
	Water Content		%	81.4
	Specific Weight		-	2.572
	Unit Weight		g/cm^3	1.106
	Grain size	Max	μ	105
		D_{60}	μ	12
		D_{30}	μ	2.4
		D_{10}	μ	-
	Liquid Limit		%	286.3
	Plastic Limit		%	130.7
	Plastic Index		-	155.6
Nutrients	Ignition Loss		%	37.6
	COD_{Mn}		mg/g	225
	COD_{Cr}		mg/g	448
	Nitrogen	T-N	mg/kg	6,290
		NH_4-N	mg/kg	7.53
		NO_2-N	mg/kg	3.22
		NO_3-N	mg/kg	3.22
	T-P		mg/kg	1,020
	Ca		mg/kg	2,540
	Sulfide		mg/g	2.28
	n-Hexane Extracts		mg/g	11.35
Toxic Substances	T-Hg		mg/kg	0.286
	Alkyl Hg		mg/kg	< 0.0005
	Pb		mg/kg	20.0
	Zn		mg/kg	34.9
	Cr		mg/kg	< 1.1
	As		mg/kg	5.81
	Cd		mg/kg	1.0
	Cyanogen		mg/kg	< 1.0
	Organic P		mg/kg	< 0.001
	PCB		mg/kg	< 0.0005

TABLE 3. DREDGE WATER RETURN FLOW STANDARDS

Property	Value	Reference
COD	20 mg/g	JIS K0102
n-Hexane Extracts	3 mg/g	JIS K0102
Hydrogen Sulfide	6700 ppm	Sewage Analysis
Total Nitrogen	625 ppm	Sewage Analysis
Total Phosphorus	88 ppm	JIS K0102

TABLE 4. PROPERTIES OF DREDGE WATER RETURN FLOWS (SUPERNATANT)

	Concentration of Sediment in Dredge Slurry			
Property	10%	20%	30%	40%
pH	7.1	7.1	7.0	6.9
SS (ppm)	17,400	29,800	52,600	83,900
Turbidity (ppm)	9,250	16,700	27,800	46,000
Transparency (deg.)	< 1.0	< 1.0	< 1.0	< 1.0
Color (deg)	52.0	67.5	86.3	110
COD$_{Mn}$ (ppm)	2,160	4,050	8,500	15,600
T-N (ppm)	67.2	143	285	582
T-P (ppm)	28.1	43.1	59.8	85.5
Sulfide (ppm)	112	265	395	677

TABLE 5. OPERATING VALUES FOR THE WATER TREATMENT PLANT

Coagulated Sedimentation				Filtration				Chlorination	Absorption by Carbon
Water Quantity	Coagulant for Pretreatment			L.V.	Reverse Washing			Sodium Hypochlorite	L.V.
	PAC	Activated Carbon Powder	Polymer		Loss Head	Washing Velocity	Time		
m^3/h	ppm	pmm	ppm	m/h	$m\text{-}H_2O$	m/h	min	ppm	m/h
3	300	100	1.0	24	5.0	60.0	5	1500-2000	6

TABLE 6. SUPERNATANT WATER QUALITIES AT EACH TREATMENT STAGE

Property	Stage			
	Input Water	Coagulated Water	Filtrated Water	Chlorinated Water
pH	7.1-8.0	6.8-7.5	6.8-7.5	6.8-8.0
Color (deg.)	500-800	50-150	20-70	20-40
Turbidity (deg.)	30-60	8-12	2-5	1-3
SS (ppm)	30-60	10-40	1-5	< 1-3
COD_{Mn} (ppm)	25-37	9-13	3-6	< 1-3
T-N (ppm)	22-25	23-24	22-24	< 1-3
NH_4-N (ppm)	20-25	22-23	22-23	< 1-3
T-P (ppm)	1.1-1.5	0.4-0.5	0.13-0.14	< 0.1
n-Hexane Extracts (ppm)	< 5	< 5	< 5	< 5
Residual Cl_2 (ppm)	---	---	---	ND
Chloramin (ppm)	---	---	---	ND
Isolated Cl_2 (ppm)	---	---	---	ND
Residual Polymer (ppm)	---	---	---	< 0.1

ND = not detectable

TABLE 7. EFFECTS OF VARYING AMOUNTS OF ACTIVATED CARBON POWDER ON THE REDUCTION OF SS AND COD

		Input Water		Added Coagulants			Coagulated Sediment		Filtration		Chlorination Absorption by Carbon	
Day		COD	SS	PAC	Carbon	PA331	COD	SS	COD	SS	COD	SS
6-10	Fluctuation	12-36	23.2-56.2	300	100	1.0	7.0-12.6	15.0-41.0	3.4-8.0	<1.0-6.0	0.4-8.0	<1.0-2.6
	Mean	24	33.0				9.7	24.3	5.5	2.0	2.3	1.3
	SS Removal						59.6	28.4	77.1	93.0	90.4	96.1
12	Fluctuation	12.0-36.0	23.2-56.2	300	50	1.0	8.0-11.0	18.8-28.0	6.0-7.2	<1.0-2.9	2.0-3.6	1.0
	Mean	24.0	33.0				9.8	22.4	5.7	1.6	2.7	41.0
	SS Removal						59.2	37.1	72.1	95.2	88.3	97.3
13-20	Fluctuation	18.0-30.0	19.5-51.5	300	0	1.0	8.0-12.5	8.0-47.0	4.0-10.4	2.8	1.2-4.4	1.8-2.2
	Mean	27.3	35.2				10.1	24.3	8.1	2.8	2.3	1.6
	SS Removal						63.0	29.6	70.3	92.1	91.6	95.4
	SS Removal						59.2-63.0	26.4-32.1	70.3-77.1	92.1-95.2	88.7-91.6	95.6-97.3
	Mean						60.6	29.4	73.2	93.7	90.3	96.2

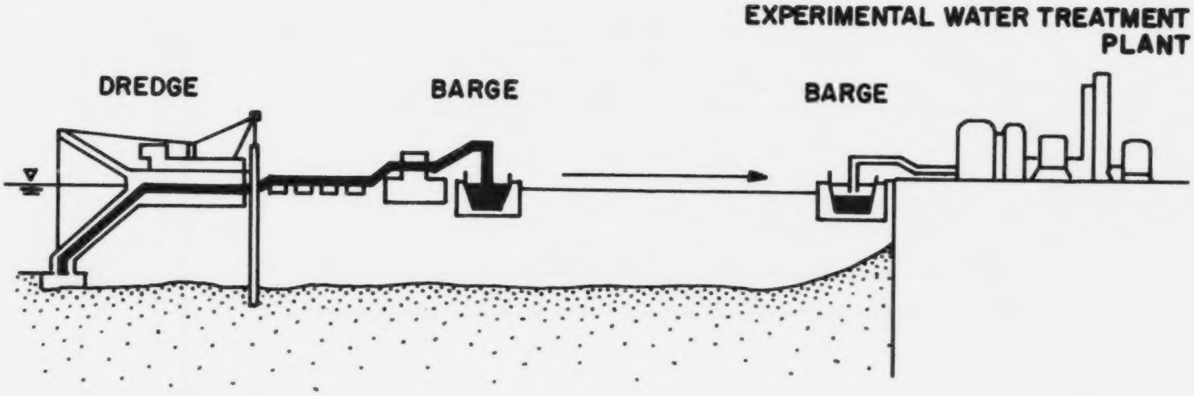

EXPERIMENTAL WATER TREATMENT PLANT
DREDGE
BARGE
BARGE

Figure 1. Schematic diagram of the experimental operation.

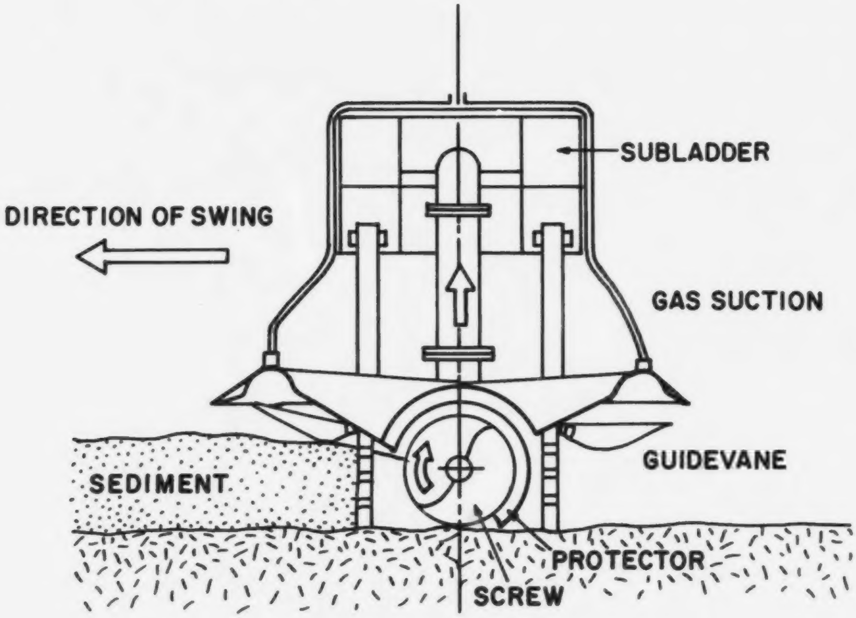

SUBLADDER
DIRECTION OF SWING
GAS SUCTION
GUIDEVANE
SEDIMENT
PROTECTOR
SCREW

Figure 2. Screw-type or auger suction head.

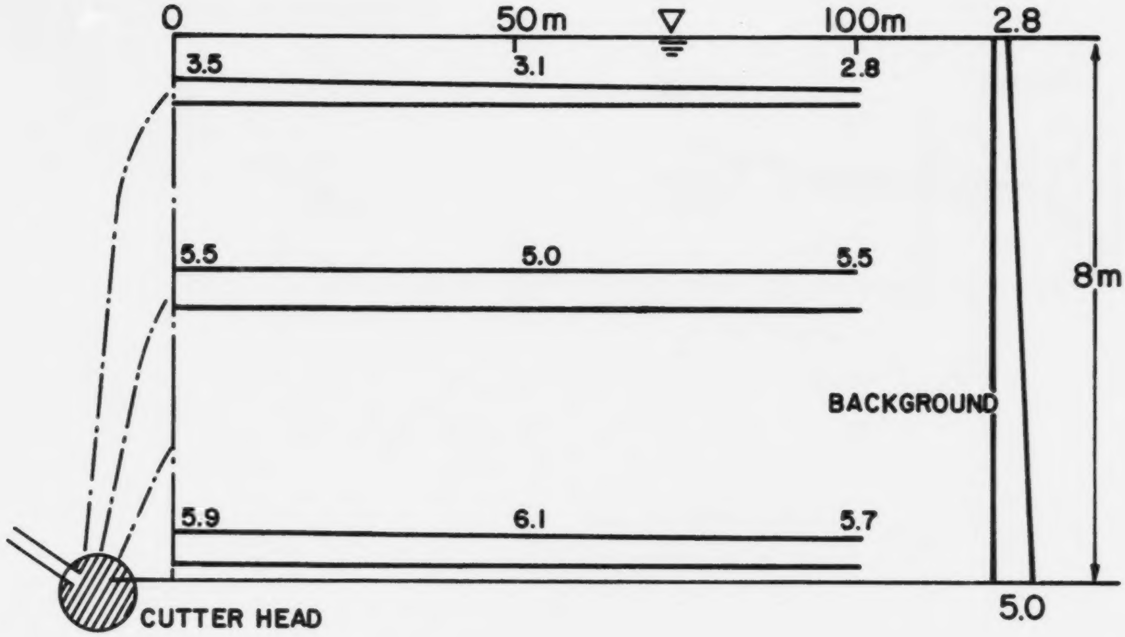

0
50m
100m
2.8
3.5
3.1
2.8
5.5
5.0
5.5
8m
BACKGROUND
5.9
6.1
5.7
CUTTER HEAD
5.0

Figure 4. Distribution of SS due to dredging (ppm).

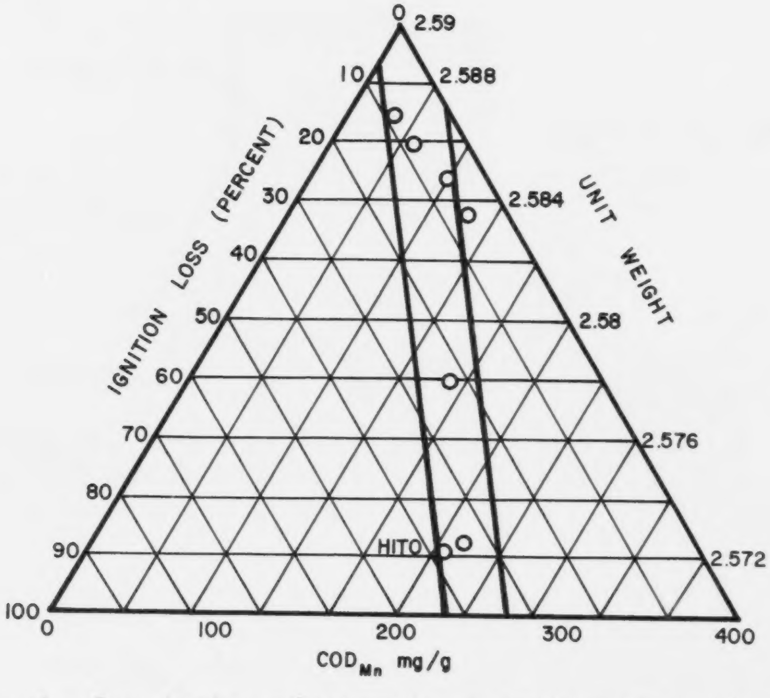


Figure 5. 3-axis plot showing relationship of ignition loss, COD and unit weight of sediment in Hiro Harbor.

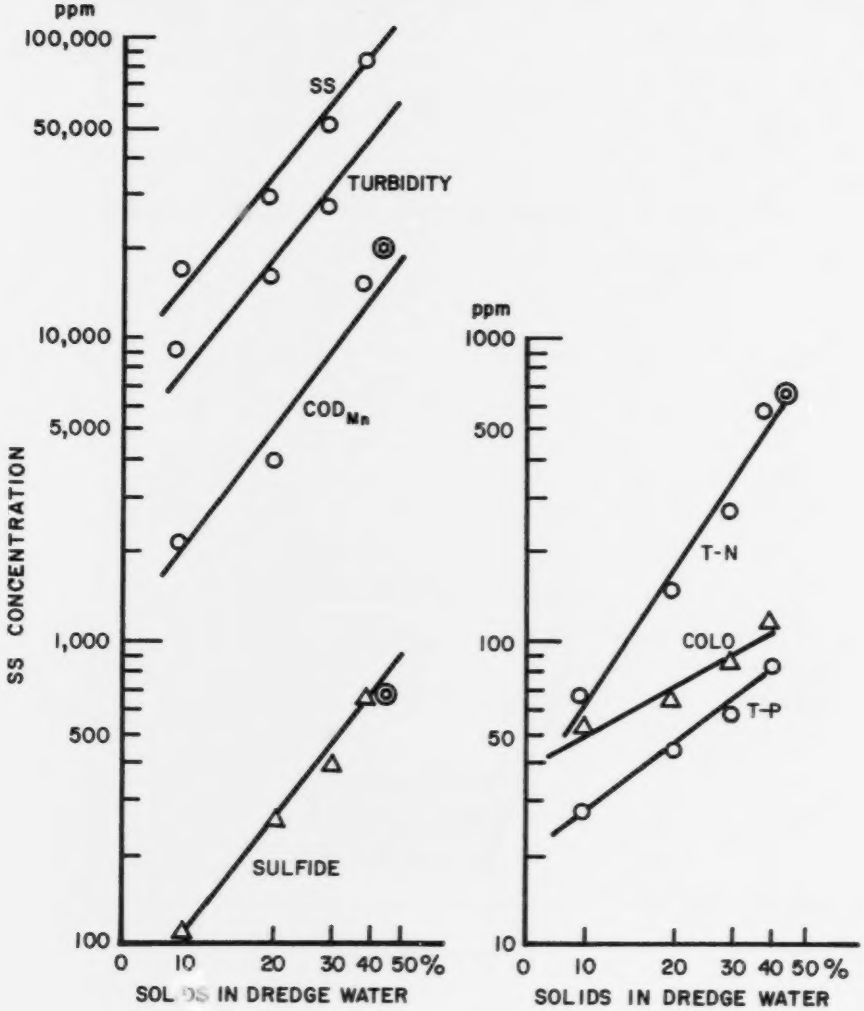


Figure 6. Relationship between concentration of various pollution components of dredge water and sediment content.

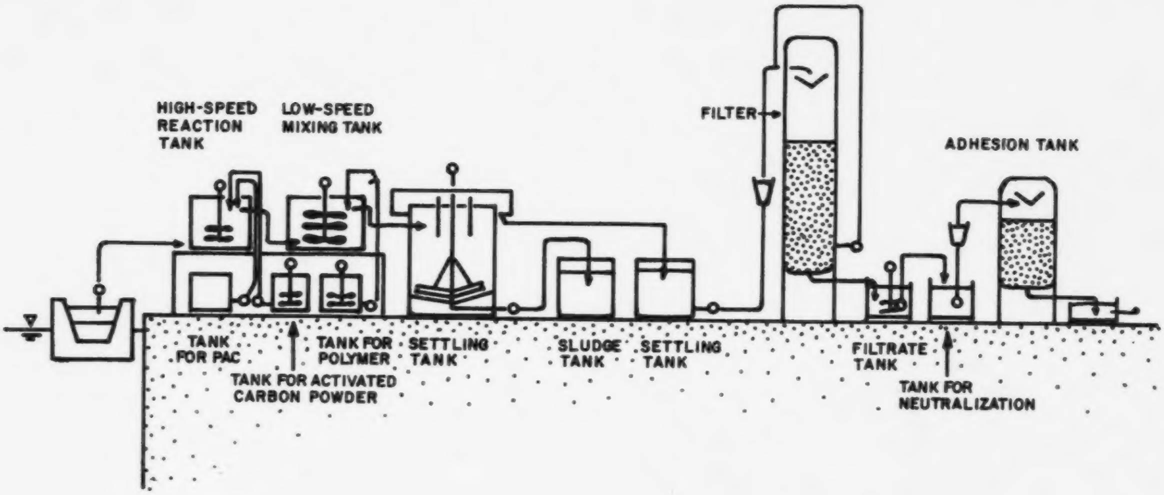
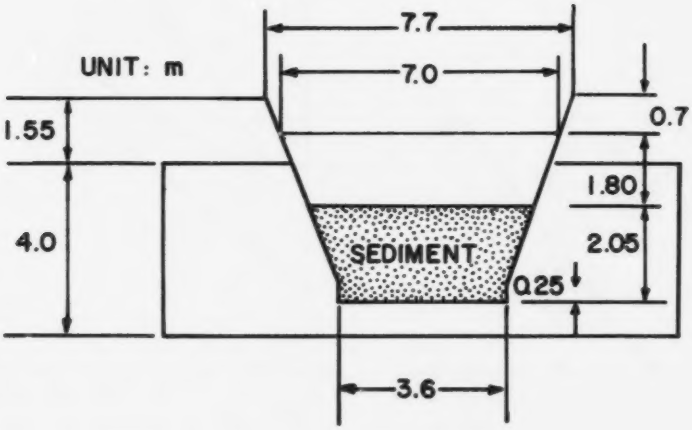

HIGH-SPEED REACTION TANK
LOW-SPEED MIXING TANK
FILTER
ADHESION TANK
TANK FOR PAC
TANK FOR POLYMER
SETTLING TANK
TANK FOR ACTIVATED CARBON POWDER
SLUDGE TANK
SETTLING TANK
FILTRATE TANK
TANK FOR NEUTRALIZATION

Figure 7. Schematic of water treatment plant.


7.7
UNIT: m
7.0
1.55
0.7
1.80
4.0
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SEDIMENT
0.25
3.6

Figure 8. Natural sedimentation in barge hold.

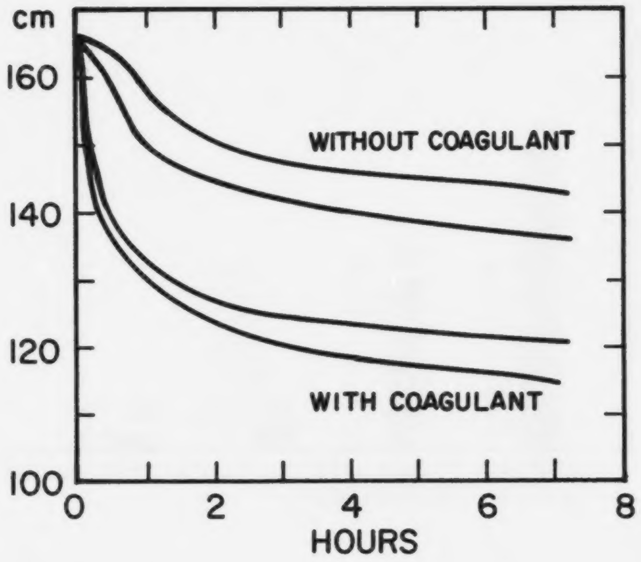

Figure 9. Precipitation curves of sediment measured by cylinder.

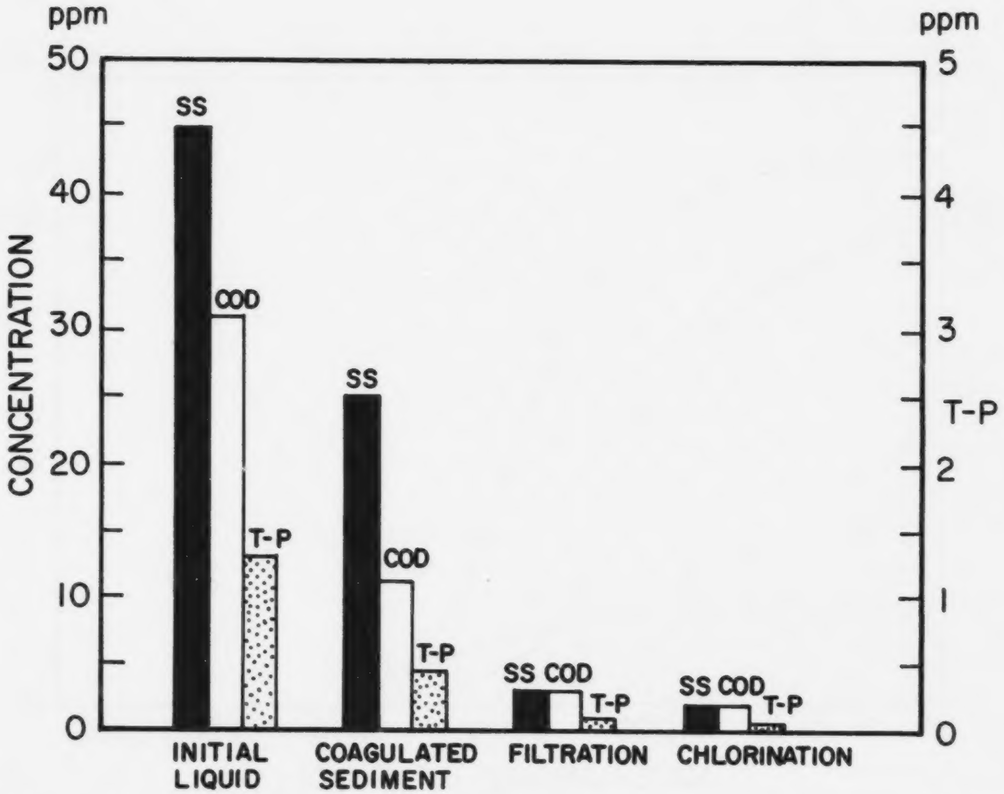

Figure 10. Reduction in concentration of SS, COD and T-P in each stage.

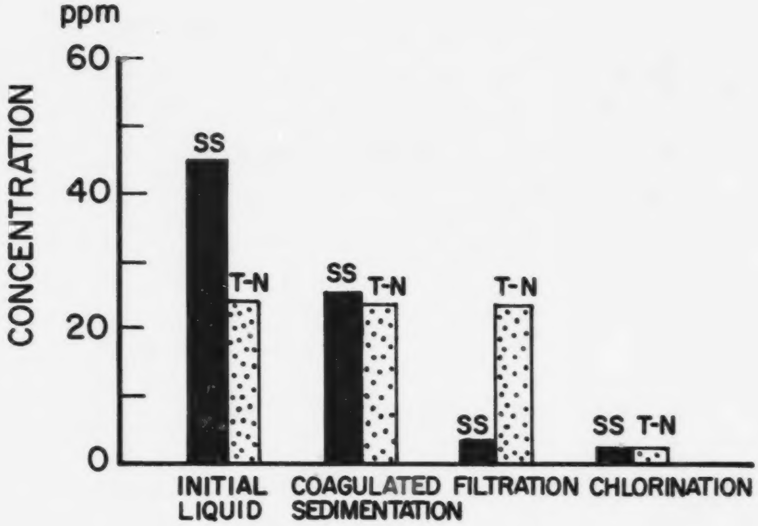

Figure 11. SS and T-N concentration at each stage.

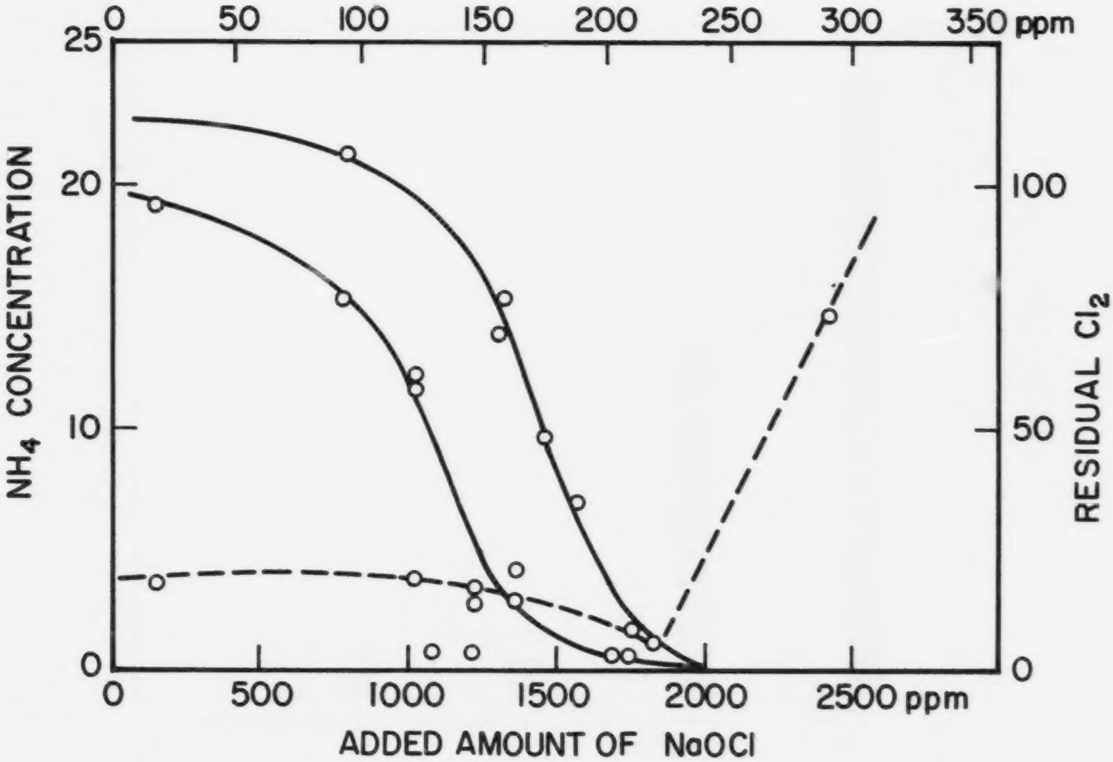

Figure 12. Relationship between NH_4 concentration and amount of NaOCl.

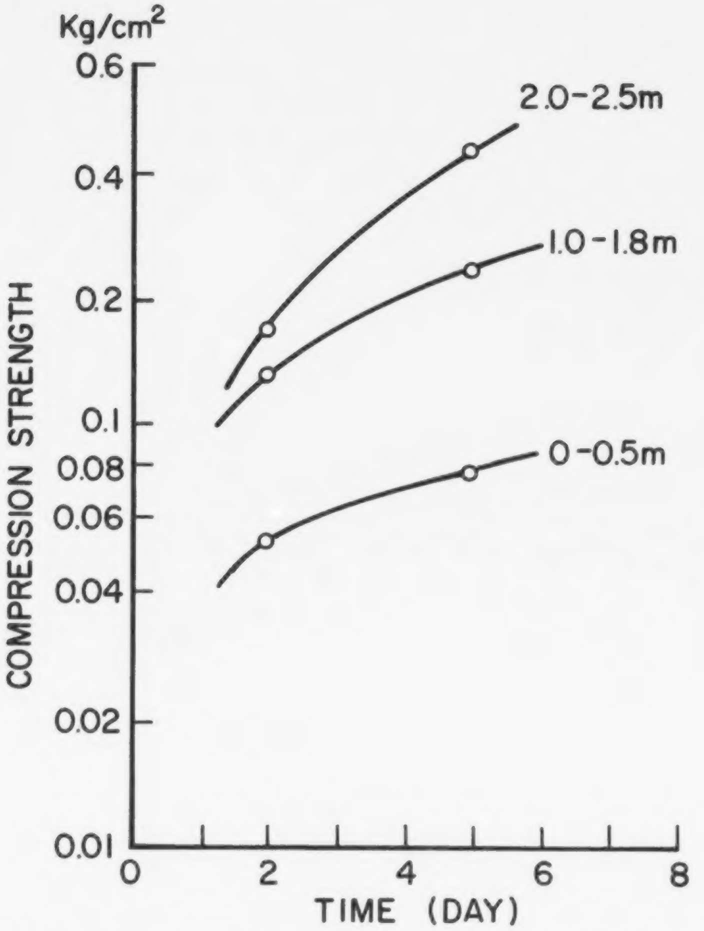

Figure 13. Compression strength of sludge after solidification.

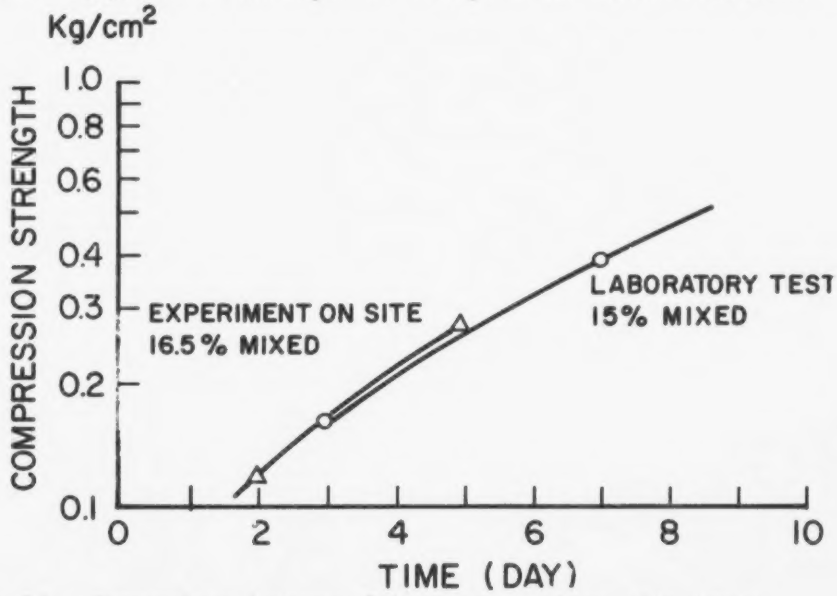

Figure 14. Comparison between laboratory and on-site tests.

THE CONTRIBUTION OF SEDIMENT TO LAKE EUTROPHICATION AS DETERMINED BY ALGAL ASSAY

Ryuichi Sudo, Chief
Mitsumasa Okada, Researcher
Freshwater Environment Laboratory
The National Institute for Environmental Studies, Japan

ABSTRACT

The contribution of sediment to lake eutrophication was determined by using the blue-green alga _Microcystis aeruginosa_, which is common in eutrophic lakes and reservoirs in Japan. Chelating agents in Gorham's medium proved necessary for the growth of _M. aeruginosa_ and it was determined that the organics in the sediments, especially fulvic acid, played the same role as the chelating agents in the growth medium.

A large release of phosphate from sediments taken from Akanoi-wan in Lake Biwa was observed during both aerobic and anaerobic incubation. The growth of _M. aeruginosa_ was highly stimulated by this material released from the sediment. Under aerobic conditions, some materials that had chelating effects were released from the surface layer (0-9 cm) of sediments at Akanoi-wan.

INTRODUCTION

The bioassay developed to determine nutrients limiting algal growth in lakes and rivers is called the "Algal Assay Procedure," abbreviated AAP (NERP, 1971). It is used to assess the effect of wastewater discharge on eutrophication. The recent increase in eutrophication in Japan has caused many researchers and adminstrators in the water pollution control field to become interested in AAP, and many reports on the application of AAP have begun to appear (Sudo _et al._, 1975; Okada and Sudo, 1978; JSWG, 1976). In almost all the previous freshwater studies that used AAP, the authors selected _Selenastrum capricornutum_ as the test alga. Because this alga has never been dominant in Japanese lakes there was a controversy over the appropriateness of its use.

Another standard test alga in AAP is _M. aeruginosa_. This alga grows in typical Japanese eutrophic lakes such as Kasumigaura and Suwako. The formation of dense blooms by this alga causes many problems including noxious odors, water treatment difficulties, and fishkills. Thus it appears desirable to use _M. aeruginosa_ as a test alga in AAP not only because it is common in Japan, but also because the reason behind the massive blooms is unknown and basic information is needed to predict their growth.

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The purpose of this study was to assess the effect of lake sediment on the growth of _M_. _aeruginosa_. This was done by using AAP to:

1) study the effects of organics in the sediment on the growth of _M_. _aeruginosa_,

2) study the effects of materials released from the sediments under both aerobic and anaerobic conditions on the growth of _M_. _aeruginosa_.

METHODS AND MATERIALS

CULTIVATION OF ALGAE

The alga used in this study was _Microcystis aeruginosa_. The test strain was obtained from the National Environmental Research Center, U.S. Environmental Protection Agency, Corvallis, Oregon. Gorham's medium was used for the stock culture medium (Table 1) (Okado and Sudo, 1978). Stationary test tube culture was satisfactory for maintaining the stock culture.

TABLE 1. COMPOSITION OF GORHAM'S NO. 11 MEDIUM

Material	Amount
$NaNO_3$	496.0 mg
$MgSO_4 \cdot 7H_2O$	75.0 mg
$CaCl_4 \cdot 2H_2O$	36.0 mg
K_2HPO_4	1.4 mg
Na_2CO_3	20.0 mg
Fe Citrate	6.0 mg
Citrate	6.0 mg
Na_2EDTA	1.0 mg
$Na_2SiO_3 \cdot 9H_2O$	58.0 mg
* Gaffron's trace elements	0.8 ml
Distilled water	999.2 ml

* Gaffron's trace elements		
H_2BO_4	3100	mg
$MnSO_4 \cdot 4H_2O$	2230	mg
$ZnSO_4 \cdot 7H_2O$	287	mg
$(NH_4)_6Mo_7O_{24} \cdot 4H_2O$	88	mg
$CO(NO_3)_2 \cdot 4H_2O$	146	mg
$Na_2WO_4 \cdot 2H_2O$	33	mg
KBr	119	mg
KI	83	mg
$Cd(NO_{32}) \cdot 4H_2O$	154	mg
$NiSO_4(NH_4)_2SO_4 \cdot 6H_2O$	198	mg
$VOSO_4 \cdot 2H_2O$	20	mg
$Al_2(SO_4)_3K_2SO_4 \cdot 24H_2O$	47	mg
1/10 N H_2SO_4	1000	ml

Precultivation of the inoculum was conducted following the same method as the algal assay. In this case, Gorham's medium was diluted to 1/5 concentration and provided as a preculture medium. This is because the high nutrient concentration in full-strength Gorham's medium may produce an inoculum that contains excess nutrient pools within the cells which would result in a maximum cell concentration and produce serious errors.

This algal assay medium was also tried as a stock and preculture medium, but the results were not satisfactory. The water samples were sterilized by filtration through a Millipore (0.45 μ) filter to prevent contamination by indigenous algae. Erlenmeyer flasks (500 ml) were used for culturing. The flasks were cleaned carefully and 100 ml quantities of water samples or media were added. A 7- to 10-day-old culture of M. aeruginosa in the exponential growth phase was washed twice and inoculated into the medium or water sample to give an initial concentration of 10,000 cells/ml. The inoculum concentration was set lower than called for by the AAP to prevent nutrient carry-over with the inoculum cells.

Cultivation was conducted in a constant temperature room (25 ± 1°C) equipped with "cool white" fluorescent lighting providing a constant light intensity of 500 lux. The flasks were set on a rotary shaker and shaken at 100 oscillations/min. Algal biomass was monitored using a Coulter Counter Model ZF equipped with a mean cell volume computer. From the cell counts and mean cell volume of the test alga, the total cell volume was calculated and this volume converted to dry weight (mg/l). Biomass measurements were continued at 2- to 3-day intervals until the biomass of alga reached a maximum.

HUMIC AND FULVIC ACID EXTRACTION FROM LAKE SEDIMENTS

Humic acid and fulvic acid were extracted from the Kasumigaura Lake sediments sampled with an Ekman dredge. A 100 g (wet weight) sediment sample was mixed with 300 ml of 0.1N NaOH and shaken 24 hrs (Figure 1). The extract was separated by filtration and adjusted to pH 3 using 0.1N HCl. After 24 hours the humic acid had precipitated and the fulvic acid still remained in the supernatant. The precipitated humic acid was washed twice with distilled water and then dissolved with 0.1N NaOH. The concentration of humic and fulvic acid were measured using Chemical Oxygen Demand.

STIMULATION OF GROWTH BY HUMIC AND FULVIC ACIDS

Gorham's medium was used as the basic medium to which was added humic and fulvic acid to assess their effects on algal growth. It is well known that these acids are natural chelating agents, so the effects of the chelating agents already contained in Gorham's medium were determined prior to the experiment.

ALGAL ASSAY ON SEDIMENT EXTRACTIONS

To assess the influence of sediment on algal growth, core samples were taken from the sediments at Akanoi-wan and Kusatsa Yamada-oki in Lake Biwa. The samples were frozen at -70°C as soon as possible after sampling, and then freeze-dried. The freeze-dried samples were mixed with lake water from the sample site which had been passed through a 1.2 μ Millipore filter. The

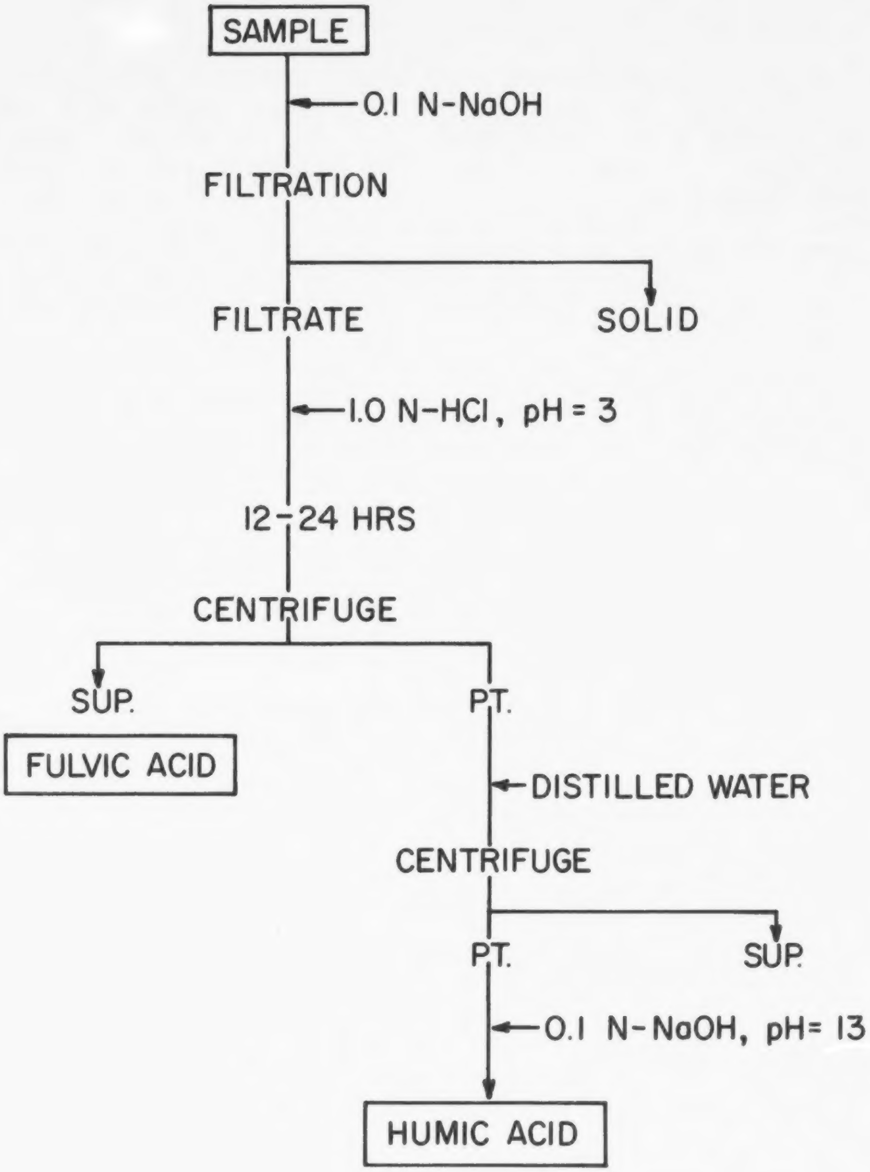
SAMPLE
0.1 N-NaOH
FILTRATION
FILTRATE
SOLID
1.0 N-HCl, pH = 3
12-24 HRS
CENTRIFUGE
SUP.
FULVIC ACID
P.T.
DISTILLED WATER
CENTRIFUGE
P.T.
SUP.
0.1 N-NaOH, pH= 13
HUMIC ACID

Figure 1. The procedure for extraction of humic and fulvic acid from sediment samples.

mixture was 1 g sediment/liter. This was transferred to a 1-liter Erlenmeyer flask and magnetically stirred and incubated in a constant temperature room held at 20 ± 1°C. An aerobic condition was maintained by aeration from the bottom of the flask with ammonia-free humidified air which had been passed through saturated boric acid and distilled water. For maintaining an anaerobic condition, the dissolved oxygen in the lake water was purged by nitrogen gas before mixing with the sediment and sealed tightly during incubation. Dissolved oxygen levels were maintained below 0.5 mg/l.

Ammonium (NH_4-N), nitrite plus nitrate (NO_2 + NO_3-N) nitrogen and phosphate (PO_4-P) concentration released from the sediment were monitored during incubation. DO, pH, and redox potential were also measured to determine if conditions were aerobic or anaerobic.

The algal assay of the sediment was conducted after 2 weeks of incubation. Any suspended solids in the mixture were separated by centrifuging and the supernatant was axenically filtered through a sterile 0.45 μ Millipore filter. Disodium ethylenediamine tetra-acetate (Na_2EDTA) at 1 mg/l, PO_4-P at 0.1 mg-P/l, and NO_3-N at 2.0 mg-N/l were added to the medium to estimate the nutrients limiting algal growth and the effect of EDTA on *M. aeruginosa*. All lake water was filtered prior to the algal assays.

RESULTS AND DISCUSSION

EFFECT OF CHELATING AGENTS ON THE GROWN OF *M. aeruginosa*

The effects of chelating agents in the Gorham's medium on *M. aeruginosa* were studied prior to adding humic and fulvic acids (Table 2). Ferric chloride or ferric citrate was the iron source at concentrations of 0.007, 0.014, and 0.028 mg-Fe/ml, and Na_2EDTA (0.3 mg/l) or citric acid (0.5 mg/l) was the chelating agent. These amounts were added to the basic medium (Table 1) without Na_2EDTA, citric acid, and ferric citrate. The values shown in Table 2 are maximum cell concentrations (mg/l) for each combination. The highest concentration was observed with the combination of ferric citrate and Na_2EDTA + citric acid. In the cases where ferric citrate was not used, *M. aeruginosa* grew with only the addition of Na_2EDTA and citric acid, but the maximum concentration was lower than with ferric citrate. It was concluded, therefore, that an iron source such as ferric citrate and a chelating agent such as Na_2EDTA were necessary for the growth of *M. aeruginosa*.

TABLE 2. EFFECTS OF IRON AND CHELATING AGENTS ON THE GROWTH OF *M. aeruginosa* [Values are maximum growth (mg/l)]

Fe conc. / Chelating Agents Conc.	$FeCl_3$ (mg/l)			Fe citrate (mg/l)		
	0.007	0.014	0.028	0.007	0.014	0.028
Na_2EDTA 0.3 mg/l	6.1	14.	7.9	-----	-----	-----
Citric Acid 0.5 mg/l	0.0	0.0	0.0	0.0	0.0	0.0
Citric Acid + Na_2EDTA	70.	38.	21.	180	160	150

EFFECTS OF HUMIC AND FULVIC ACIDS ON THE GROWTH OF *M. aeruginosa*

Humic and fulvic acids are natural chelating agents which may stimulate the growth of *M. aeruginosa*. Figure 2 shows the growth curve for *M. aeruginosa* when grown in the basic medium (including chelating agents). The addition of each acid increased the maximum specific growth rate from 0.25/day to 0.4/day. In the instance where 12.5 mg/l fulvic acid was added, maximum cell concentration increased as well. The lag phase was prolonged when humic acid was added and the maximum cell concentration was not much different from that grown on the basic medium.

Figure 3 shows the results of a similar experiment except that no chelating agent was used in the basic medium. In all cases, the specific growth rate of *M. aeruginosa* was decreased, and without either humic or fulvic acids the growth rate of the control was only 0.1/day. With the addition of fulvic acid at 12.5 mg/l, the growth rate doubled to 0.22/day and the maximum cell concentration at 34 mg/l was three times that of basic medium (11 mg/l). The humic acid sample (7.3 mg/l) also showed a higher maximum cell concentration of 17 mg/l.

These results show that both fulvic and humic acid could stimulate the growth of *M. aeruginosa*. Fulvic acid had the most effect. These effects are not as pronounced as those produced by the chelating agents used in Gorham's medium.

THE EFFECTS OF WATER-EXTRACTIONS OF SEDIMENT ON THE GROWTH OF *M. aeruginosa*

Core samples taken at Akanoi-wan were separated into four subsamples by depth: 0-4 cm, 4-9 cm, 9-14 cm, 14-24 cm. Each sample was incubated under both aerobic and anaerobic conditions and monitored for the release of nitrogen and phosphorus.

The concentration of PO_4-P released from the 4 different subsample types during a 55-day incubation period is shown in Figures 4 and 5. Under both aerobic and anaerobic conditions, the PO_4-P concentration in the upper layer was higher than that of the lower layer. The PO_4-P concentration reached a maximum of 0.8 mg P/l for the aerobic experiment and 1.7 mg P/l for anaerobic conditions over 25 days. A pH of 8.0 and redox potential of 410 mV were maintained throughout the aerobic experiment. Under anaerobic conditions, a lower pH of 6.8-7.5 was observed and the redox potential decreased from 400 mV to 100 mV after a day and remained there during the rest of the incubation period.

The effect of different incubating conditions on PO_4-P concentration is shown in Figures 6 and 7. PO_4-P concentrations under anaerobic conditions were higher than under aerobic conditions. The PO_4-P concentration at the time of sampling at the bottom and top of the water column was 0.14 mg P/l and 0.03 mg P/l respectively, therefore it is likely that the sediments have a high potential for releasing phosphate under both aerobic and anaerobic conditions.

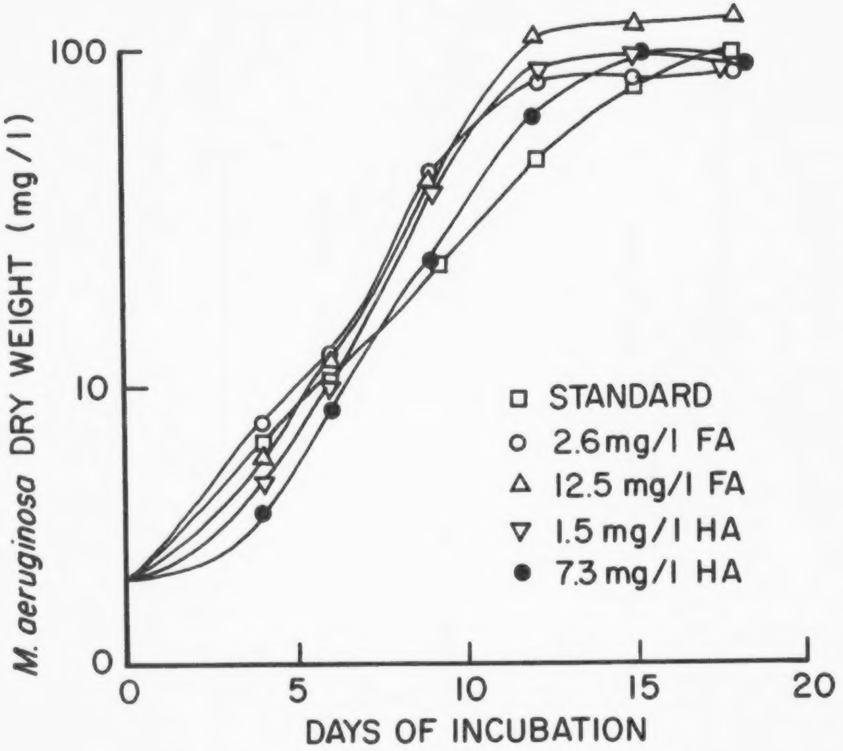

Figure 2. Effect of humic acid (HA) and fulvic acid (FA) on the growth of M. aeruginosa cultivated in Gorham's Meduim No. 11 containing chelating agents.

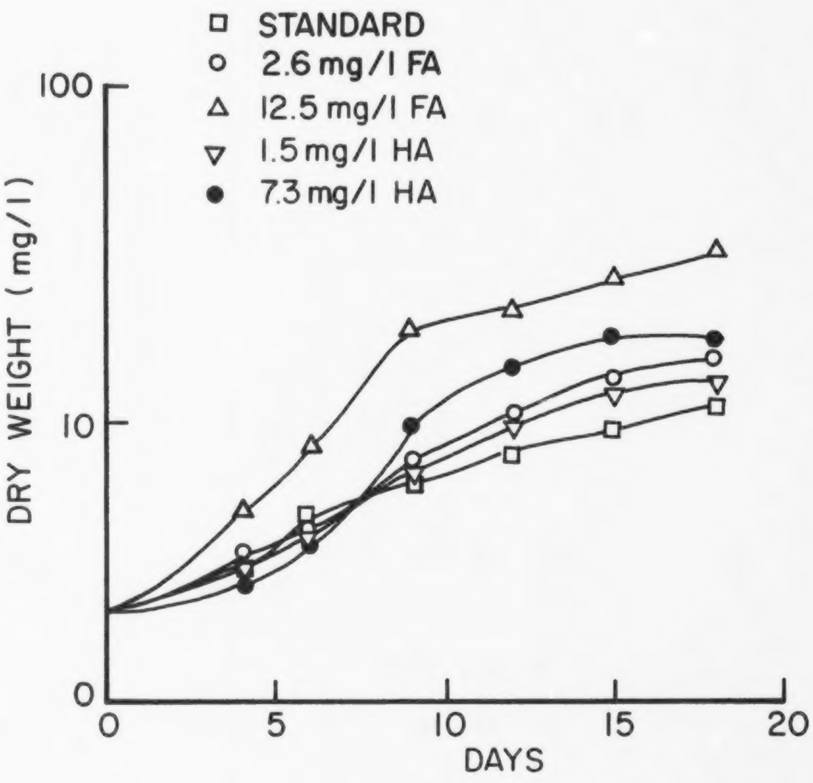

Figure 3. Effect of humic acid (HA) and fulvic acid (FA) on the growth of M. aeruginosa cultivated in Gorham's Meduim No. 11 but excluding the chelating agents.

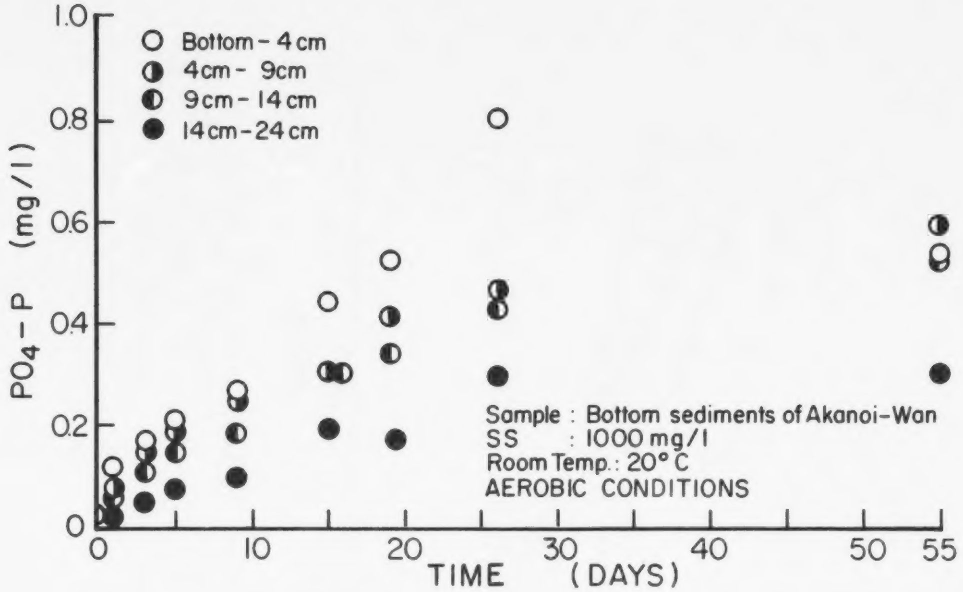

Figure 4. Concentration of PO_4-P during aerobic incubation of lake sediment from Akanoi-wan.

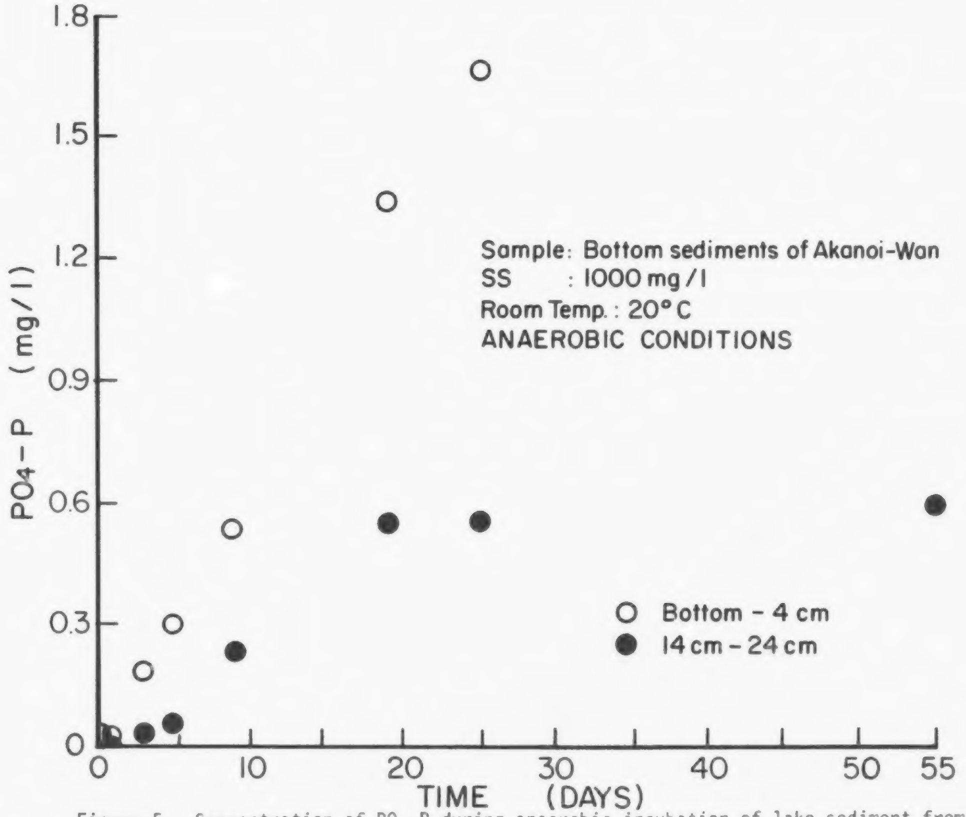

Figure 5. Concentration of PO_4-P during anaerobic incubation of lake sediment from Akanoi-wan.

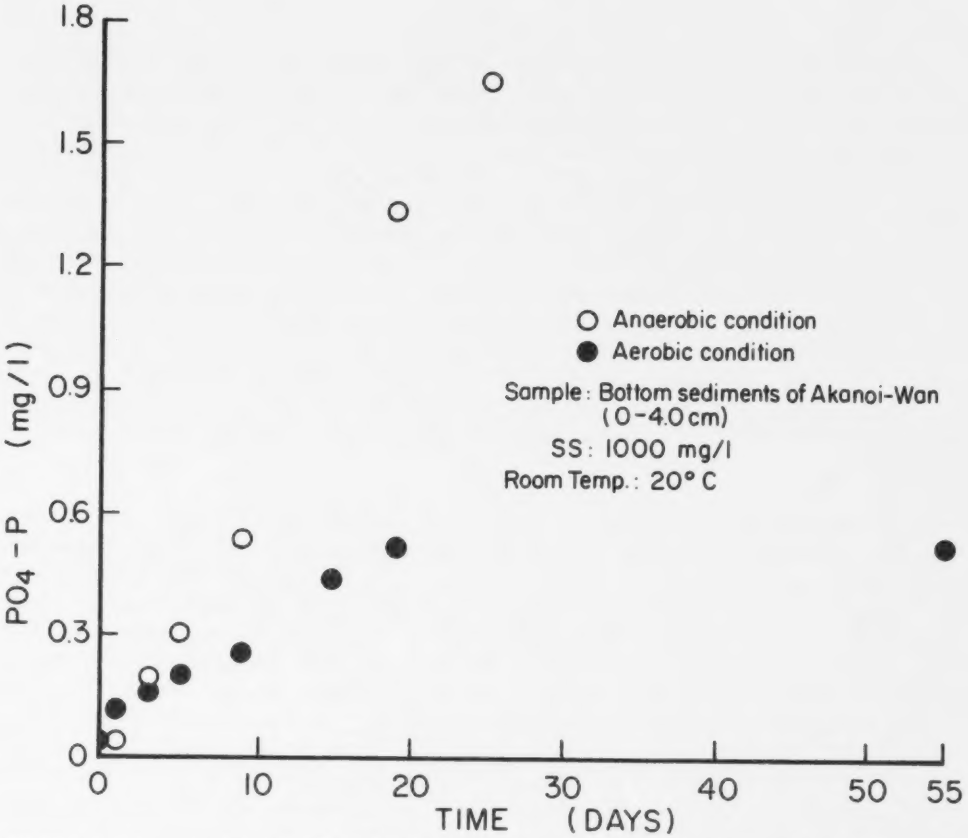

Figure 6. Concentration of PO_4-P during aerobic and anaerobic incubation of lake sediment from Akanoi-wan 0-4 cm layer.

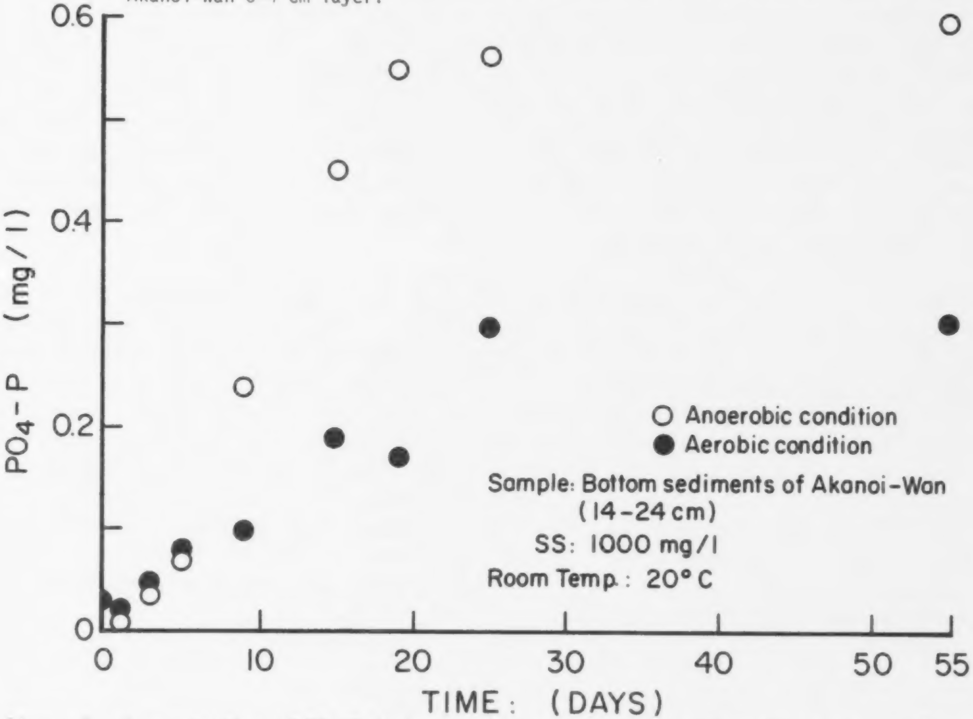

Figure 7. Concentration of PO_4-P during aerobic and anaerobic incubation of lake sediment from Akanoi-wan 14-24 cm layer.

Figures 8 and 9 show the release of NH_4-N and NO_2 + NO_3-N from the sediments at core depths of 0-4 cm and 14-24 cm. Under anaerobic conditions a high concentration of NH-N was released, but NO_2 + NO-N was not detected until after a day of incubation. This means that denitrification occurred in the incubator which suggests that mainly NH_4-N would be released from the sediments under anaerobic conditions. But NO_2 and NO_3-N were found in the aerobic incubator. NH_4-N was detected only at the beginning of the incubation period and then decreased due to nitrification. Although the forms of nitrogen released were different under aerobic and anaerobic conditions, the amount (1.2 mg N/l) of total inorganic nitrogen (NH_4 + NO_2 + NO_3-N) did not change except in the aerobic incubation of the 14-24 cm core sample. It is generally accepted that the nitrogen available for the growth of algae is inorganic nitrogen, therefore the fact that total available nitrogen was the same under all conditions differs from the case of phosphate, where the type of ion is significant.

Another sampling station at Kusatsu Yamada-oki is not as polluted as Akanoi-wan. Sediment from this station was separated into three subsamples by depth in the core: 0-2.5 cm, 2.5-7.5 cm, 7.5-12.5 cm. The sediment in this area was grey and different from the black sediment at Akanoi-wan. The PO_4-P concentration released under the two incubation conditions from sediments at different depths is shown in Figures 10 and 11. As at Akanoi-wan, more PO_4-P was released from the upper layer, but the total amount released from the upper layer was one-half of that at Akanoi-wan. In the 7.5-12.5 cm layer, the PO_4-P concentration was lower than 0.1 mg P/l even under anaerobic conditions. It is possible that this is because suspended solids with high phosphorus concentrations have been settling for a long time at Akanoi-wan but only recently at Kusatsu Yamada-oki.

Figures 12 and 13 show the PO_4-P released from the 0-2.5 cm and 7.5-12.5 cm layers under different incubation conditions. In the 7.5-12.5 cm layer, anaerobic release was higher than aerobic release. But, there was little difference between aerobic and anaerobic conditions in the 0-2.5 cm layer. This result is very different from that at Akanoi-wan. Although the concentration of PO_4-P released from the Kusatsu Yamada-oki sediment was very low, the potential for sediment phosphorus released is high due to the low PO_4-P concentration in lake water; i.e. the bottom had 0.21 mg P/l and the lake surface had 0.001 mg P/l as PO_4-P.

Figures 14 and 15 show the amounts of inorganic nitrogen released from the 0-2.5 cm and 7.5-12.5 cm sediment layers. Under both aerobic and anaerobic conditions, the release of NH_4-N was remarkable. As at Akanoi-wan, the maximum concentration was higher under anaerobic conditions; i.e. 0.76 mg N/l for the 0-2.5 cm layer and 0.5 mg N/l for the 7.5-12.5 cm layer. In the case of PO_4-P, there was a large difference between Akanoi-wan and Kusatsu Yamada-oki; however, the amount of nitrogen released from Kusatsu Yamada-oki was roughly half that of Akanoi-wan. The results under aerobic conditions show that nitrification did not occur in the Kusatsu Yamada-oki sediment and that the NO_2 + NO_3-N concentration was always low. This is noticeably different than the case at Akanoi-wan. As at Akanoi-wan, the depth of sediment samples made only a small difference.

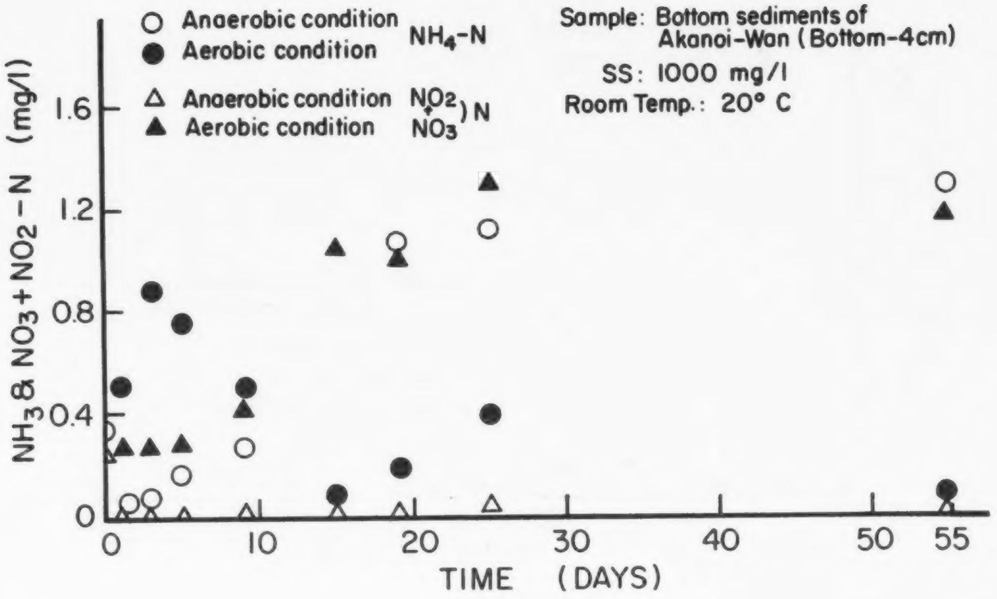

Figure 8. Concentration of NH_4-N and NO_2 + NO_3-N during aerobic and anaerobic incubation of lake sediments from Akanoi-wan 0-4 cm layer.

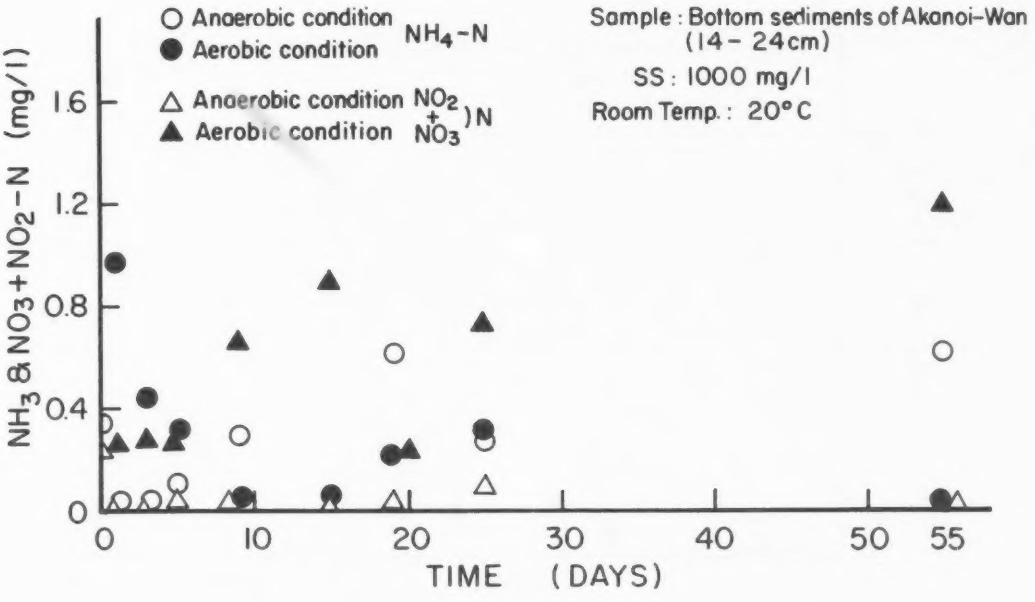

Figure 9. Concentration of NH_4-N and NO_2 + NO_3-N during aerobic and anaerobic incubation of lake sediment from Akanoi-wan 14-24 cm layer.

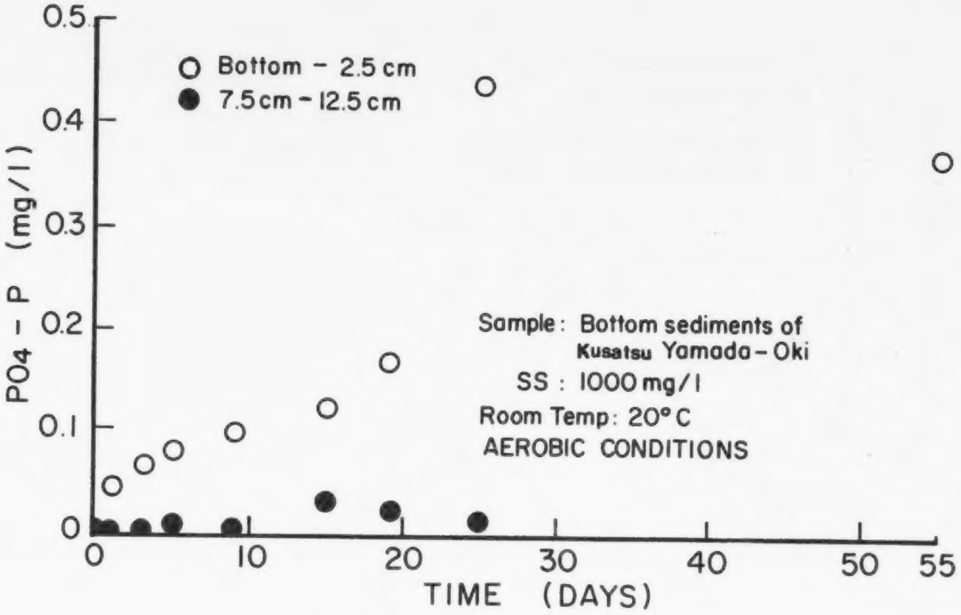

Figure 10. Concentration of PO_4-P during aerobic incubation of lake sediment from Kusatsu Yamada-oki.

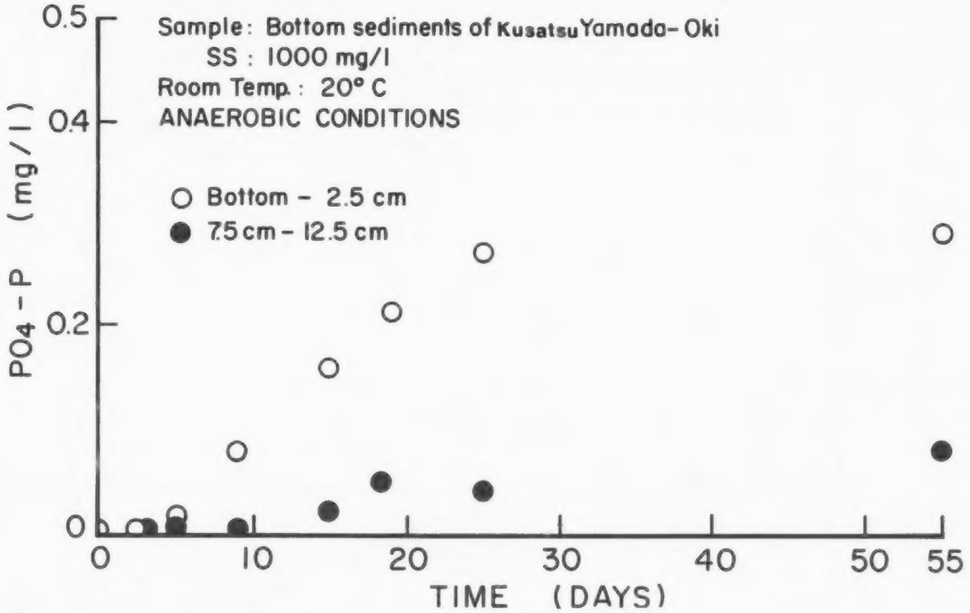

Figure 11. Concentration of PO_4-P during anaerobic incubation of lake sediment from Kusatsu Yamada-oki.

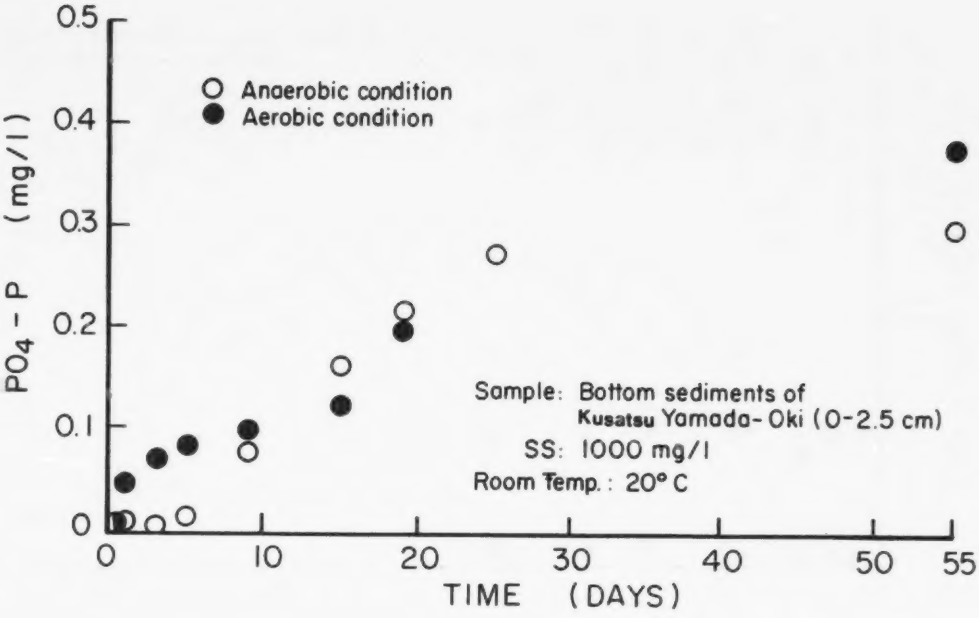

Figure 12. Concentration of PO_4-P during aerobic and anaerobic incubation of lake sediment from Kusatsu Yamada-oki 0-2.5 cm layer.

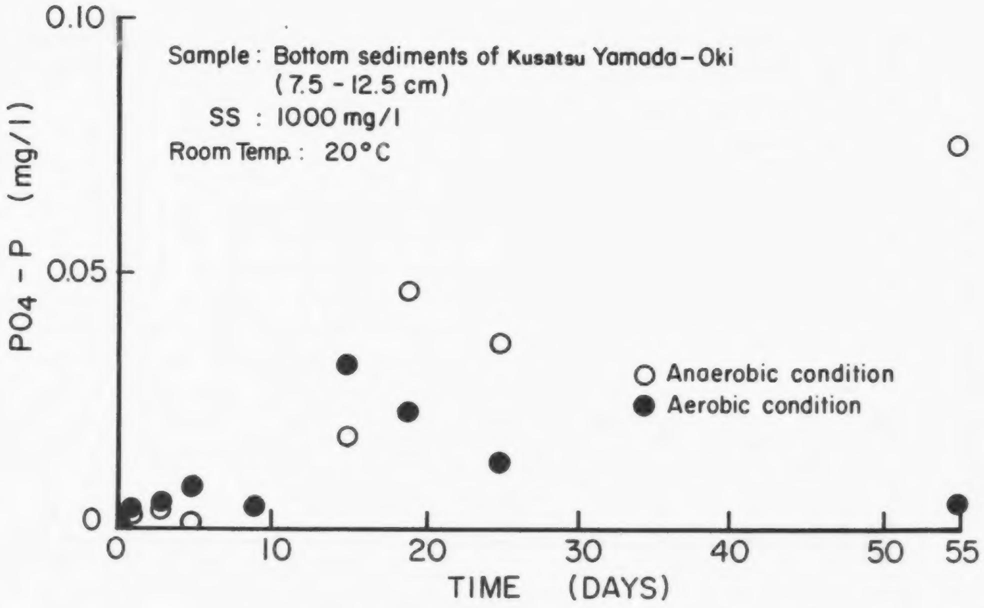

Figure 13. Concentration of PO_4-P during aerobic and anaerobic incubation of lake sediment from Kusatsu Yamada-oki 2.5-7.5 cm layer.

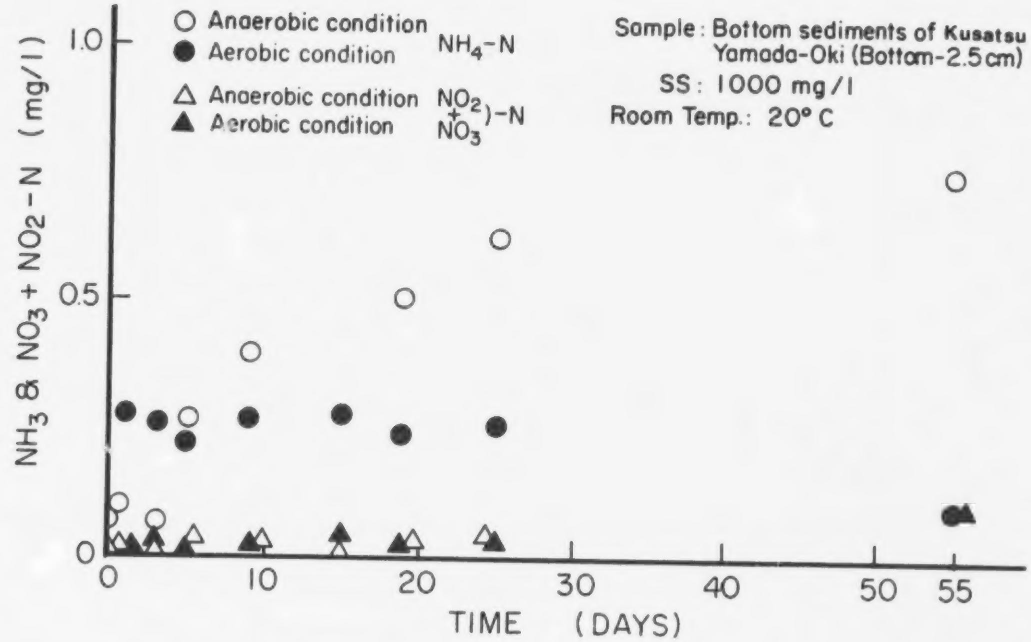

Figure 14. Concentration of NH_4-N and NO_2 + NO_3-N during aerobic and anaerobic incubation of lake sediment from Kusatsu Yamada-oki 0-2.5 cm layer.

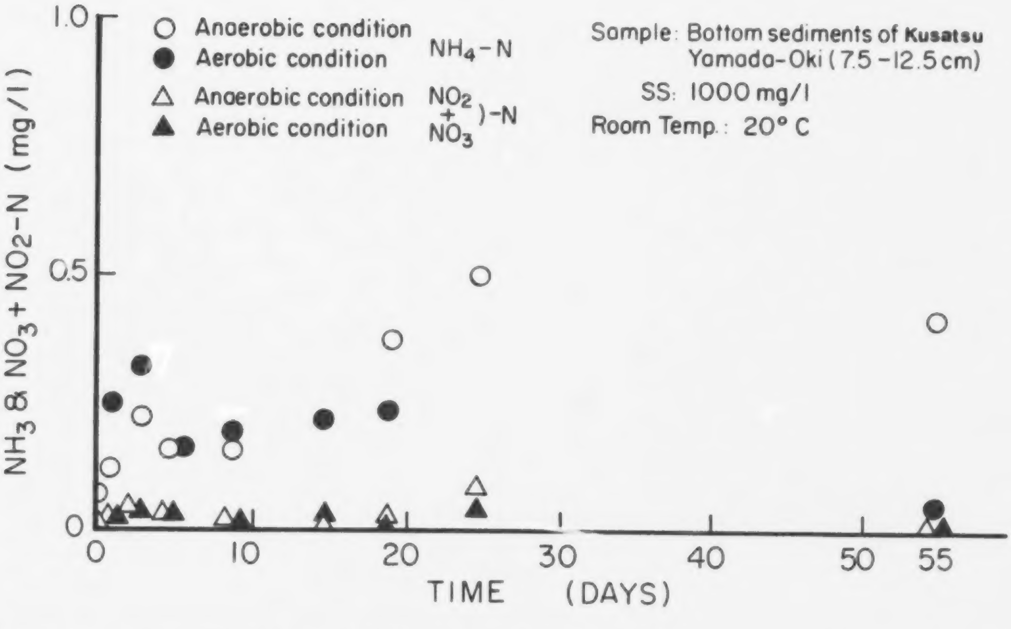

Figure 15. Concentration of NH_4-N and NO_2 + NO_3-N during aerobic and anaerobic incubation of lake sediment from Kusatsu Yamada-oki 7.5-12.5 cm layer.

A 14-day-old sediment-lakewater extract was used for the algal assays. Because the PO_4-P concentration was very high, a dilution technique was adopted to assess the algal growth potential (AGP) of the sample water. AGP was calculated using the following equation:

AGP (mg/l)= maximum cell concentration (mg/l) x dilution ratio.

The nutrient concentration and AGP of samples from Akanoi-wan are shown in Table 3 and Figure 16. AGP in lake water increased from 6.9 mg/l to 15.2 mg/l by introduction of EDTA. A nutrient pulse test showed higher AGP values than in the case of the nitrogen pulse test, but it is difficult to conclude that nitrogen was the limiting nutrient.

TABLE 3. NUTRIENT CONCENTRATIONS IN AKANOI-WAN WATER SAMPLES USED FOR ALGAL ASSAY

		Total P	PO_4-P	NH_4-N	NO_2+NO_3-N
Filtered lake water		0.029	0.027	0.034	0.025
0.0-0.4 cm :	Oxic	0.510	0.440	0.08	1.06
0.0-4.0 cm :	Anoxic	2.30	1.83	0.09	0.00
4.0-9.0 cm :	Oxic	0.496	0.304	0.05	0.89
9.0-14.0 cm :	Oxic	0.394	0.304	0.08	0.88
14.0-24.0 cm:	Oxic	0.261	0.190	0.06	0.90
14.0-24.0 cm:	Anoxic	0.738	0.452	0.05	0.00

The AGP was high in the anaerobic water extract from the upper layer (0-4 cm, 4-9 cm). This seems to be caused by the high concentration of nitrogen and phosphate released from the sediment. However, the AGP of the lower layer (9-14 cm, 14-24 cm) was very low even though the nutrient concentrations were high. By adding EDTA, the AGP of the lower layer increased to the level of the upper layer without increasing the AGP of the upper layer. It is possible that materials similar to EDTA released nutrients in the upper layer, but did not release them from the lower layer or did release some kind of algal growth inhibiting materials. A nutrient pulse test showed that the amount of phosphate released was much greater than expected based on the N/P ratio required for the growth of M. aeruginosa.

An aerobic sediment extract showed very low AGP in both the upper and lower layers. This result seems to be caused by a low nitrogen concentration. The same results were obtained from the nutrient pulse test, i.e. AGP in nitrogen spiked samples was very high.

The nutrient concentration in water samples from Kusatsu Yamada-oki are shown in Table 4 and the AGP is shown in Figure 17. The AGP of sediment extract with a pulse of EDTA added was higher than that without EDTA. The growth stimulation effect that was observed in the sediment from Akanoi-wan did not appear. Although the concentration of PO_4-P was lower than that of Akanoi-wan, the PO_4-P released from Kusatsu Yamada-oki caused a high AGP value where a nitrogen pulse was added.

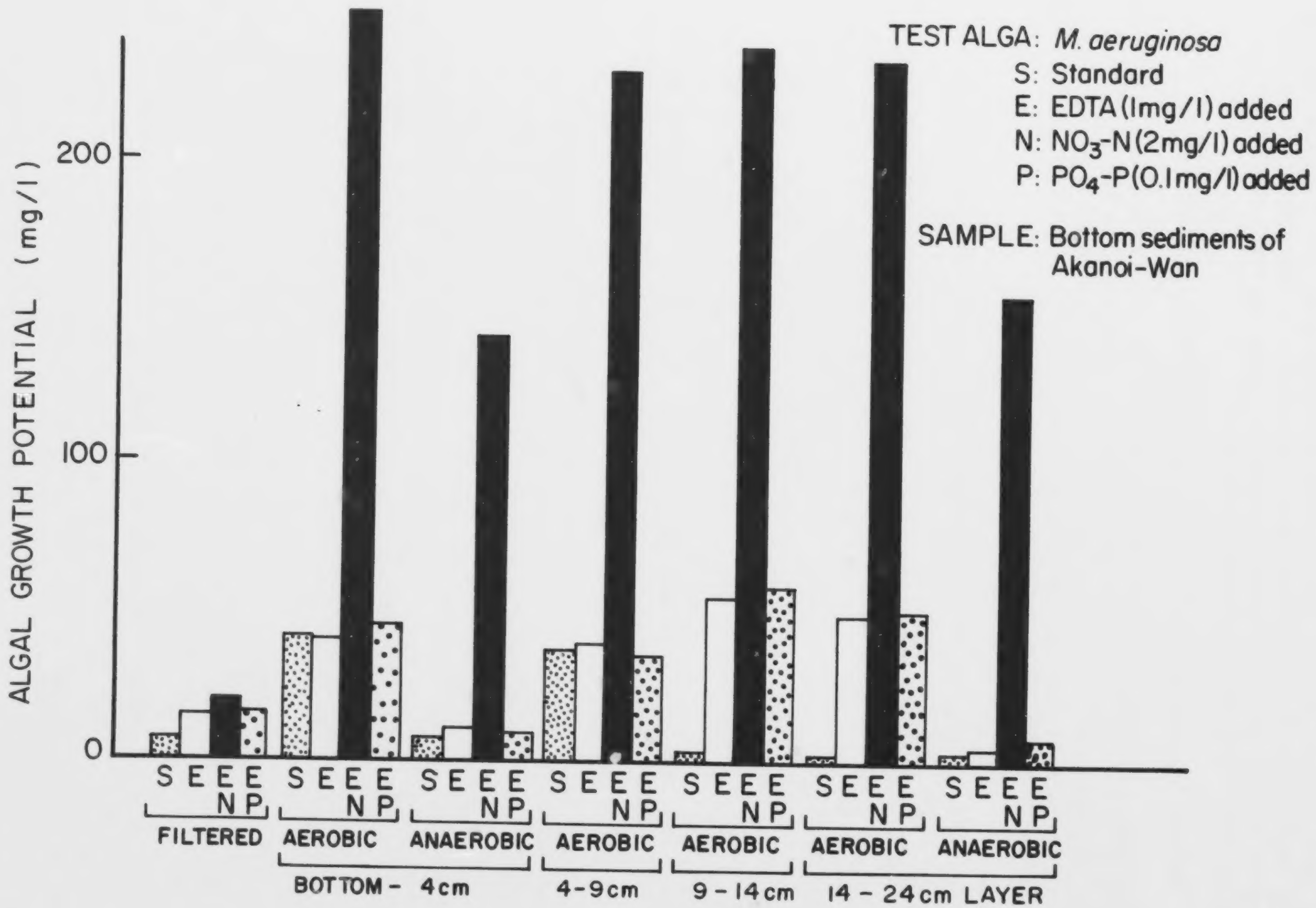

Figure 16. Algal growth potential of lake water and sediment extract (Akanoi-wan).

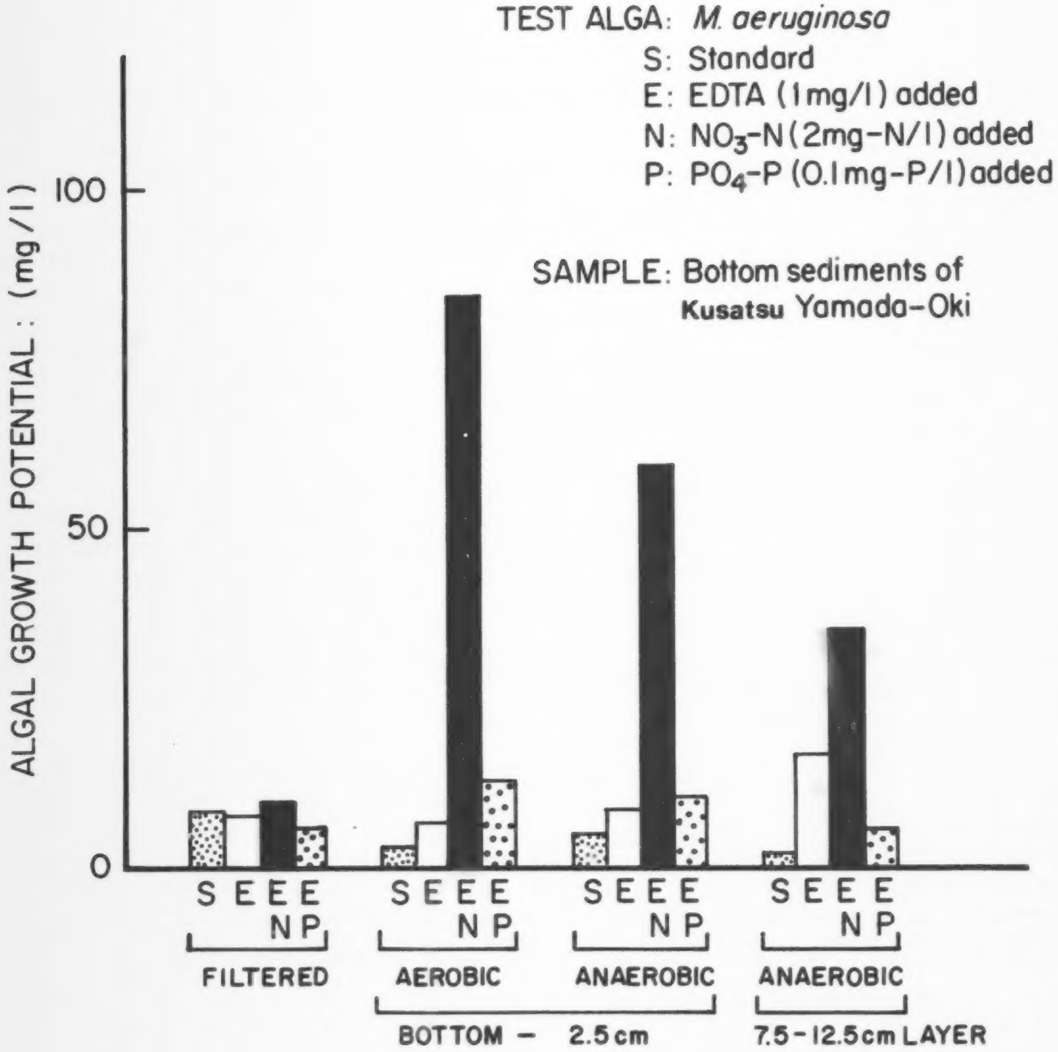

Figure 17. Algal growth potential (AGP) of lake water and sediment extracts from Kusatsu Yamada-oki.

TABLE 4. NUTRIENT CONCENTRATIONS IN KUSATSU YAMADO-OKI WATER SAMPLES USED FOR ALGAL ASSAY

	Total P	PO_4-P	NH_4-N	NO_2+NO_3-N
Filtered lake water	0.001	0.000	0.07	0.01
0.0-2.5 cm : Oxic	0.132	0.122	0.28	0.04
0.0-2.5 cm : Anoxic	0.199	0.159	0.12	0.00
7.5-12.5 cm: Anoxic	0.032	0.018	0.08	0.01

THE EFFECTS OF SEDIMENTS ON THE GROWTH OF M. aeruginosa

Chelating agents such as EDTA and ferric citrate play an important role in the growth of M. aeruginosa. The origin of these chelates in natural water is in the sediment or metabolites from algae, bacteria, and other organisms. In this research, fulvic acid extracted during the spring from lake sediment showed a growth stimulation effect; a more remarkable effect was observed when fulvic acid was extracted from highly polluted sediments in summer. Therefore, it can be concluded that the sediments play an important role in the growth of M. aeruginosa.

Algal assays were conducted on sediments sampled in Lake Biwa to clarify the differences between sampling stations and sediment depth. Higher AGP values were observed in the polluted Akanoi-wan sediment extracts compared to the less polluted Kusatsu Yamada-oki sediment extracts. This difference resulted not only from the large amount of nutrients released from the sediment at Akanoi-wan, but also from the release of materials equivalent to chelates. Since these chelating agents were not released under anaerobic conditions, the polluted sediment appears to stimulate the growth of M. aeruginosa even under aerobic conditions.

Advanced wastewater treatment and nutrient capture or diversion are regarded as effective methods for lake reclamation. However, not all of these techniques result in success because of internal phosphorus loads (Welch, 1977). It is therefore important to prevent the release of nutrients and organic materials not only to reduce internal nutrient loads, but also to decrease the concentration of materials which stimulate algal growth, particularly for the troublesome blue-green algae, M. aeruginosa.

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TOXIC MATERIAL AND NUTRIENTS FROM CONTAMINATED SEDIMENTS

Yoshiharu Nakazono and Yasuji Saotome
Machinery Division
Port and Harbor Research Institute
Ministry of Transport, Japan

ABSTRACT

This paper describes the characteristics of PCB (polychlorinated biphenyls), mercury, phosphate-P, and nitrogen from contaminated sediments. It discusses the adherence of these materials to soil particles, their removal by coagulants and a sand filter, and their solubility in water.

The smaller the soil particle size, the higher the content of PCB or mercury. Phosphate-P can be removed by the addition of certain coagulants, but mercury, PCB, and nitrogen cannot be removed by the coagulating agents used in this work. PCB and mercury are readily removed by suspended solids.

INTRODUCTION

Contaminated bottom sediments, for reasons of pollution control and environmental preservation, must be removed as soon as possible. Removal is currently done by using conventional dredging and reclamation systems. These techniques require great care to avoid secondary pollution such as dredging-caused turbidity or wastewater discharged from a reclamation site. Various dredges and reclamation methods have been designed to deal with these problems. Sediments removed and sent to reclamation sites must be enclosed by bulkheads, and wastewater discharged to the sea must meet standards set by the Cabinet Order for Implementation of the Marine Pollution Prevention Law. These are shown in Table 1. The removal standards for sediments containing mercury and PCB are established by the government. In accordance with these criteria, sediments containing more than 25 ppm mercury in rivers and marshes are subject to removal, and the ocean criteria are determined by the mean range, solubility ratio of mercury from the sediments, and the safety factor relative to any fisheries in the area. Any bottom sediments with more than 10 ppm of PCB should be removed.

Red tides will result from eutrophication caused by the discharge of wastewater from industrial plants or by the dumping of domestic wastewater, which provides nutrients such as nitrogen and phosphate-P.

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TABLE 1. EFFLUENT STANDARDS (June 21, 1971)

Toxic Substances	Permissible Limits (mg/l)
Cadmium and its compounds	0.1
Cyanide compounds	1
Organic phosphorus compounds (parathion, methyl parathion, methyl demeton and EPN only)	1
Lead and its compounds	1
Chromium (VI) compounds	0.5
Arsenic and its compounds	0.5
Total mercury	0.005
Alkyl mercury compounds	Not detectable[1]
PCB	0.003

[1] "Not detectable" means that the substance is below the level detectable by the standard method designated by the Director General of the Environmental Agency.

STUDY METHODS AND PROCEDURES

SEDIMENT SAMPLING

The contaminated sediments used in this study were sampled at Tagonoura Port for PCB and nitrogen, at Nagoya Port and Tokyo Bay for mercury, and at the mouth of the Tama River for phosphate-P. PCB at Tagonoura Port originated in the wastewater from paper mills which used to produce paper containing PCB. Straw chips and coarse rubble were removed from the samples according to the standard method. Samples were collected with Koken and Kumada-type samplers. The former is a grab sampler and the latter is a dredge sampler. The samples were packed in polyethylene bags. Their identification and conditions are shown in Table 2.

In this study the following tests were performed:

1) Sedimentation Test
2) Filtration Test
3) Solubility Test

Concentrations were determined for PCB, total mercury, phosphate-P and total nitrogen according to standard methods. It is not necessary to be concerned about detection limits in soil samples because of the high concen-

TABLE 2. SAMPLES AND SAMPLING CONDITIONS

Sample	Date	Conditions of Sediments	Depth (m)	Remarks
PCB-1	Nov. 1976	black, hydrogen sulfide smell	9.0	
PCB-2	"	" " " "	12.0	
PCB-3	"	" " " "	15.0	
Hg-1	Nov. 1977	" " " "	2.1	
Hg-2	"	" " " "	2.0	mixture
P-1	Oct. 1976	"	3.0	
P-2	"	black, sandy	4.9	
P-3	"	black, clayey	25.0	
N	Nov. 1977	black, hydrogen sulfide smell	6.5	

trations, but in water the detection limits are as follows for the instruments and the sample volumes used:

1) PCB -- 0.0005 mg/l
2) Total Mercury -- 0.0001 mg/l
3) Phosphate-P -- 0.003 mg/l
4) Total Nitrogen -- 0.5 mg/l

In performing these tests, both natural Tokyo Bay and seawater and artificial seawater made from "Aquamarine" were employed. The natural seawater was used for the PCB and phosphate-P analyses, and the artificial seawater for the total mercury and total nitrogen analyses.

SAMPLE PROPERTIES

Samples were analyzed for true specific gravity, grain size distribution, ignition loss, and the content per weight of dried sediment for the items of interest. The results shown in Table 3 and Figure 1 demonstrate that every sample consists primarily of particles below 74 μm and that the true specific gravity is comparatively low because of the high ignition loss. All the samples were prepared by centrifuging at 3,000 rpm for 20 minutes according to the standard method, and testing the resultant precipitate.

The preliminary removal standard for sea bottom sediments containing mercury is calculated using the following equation:

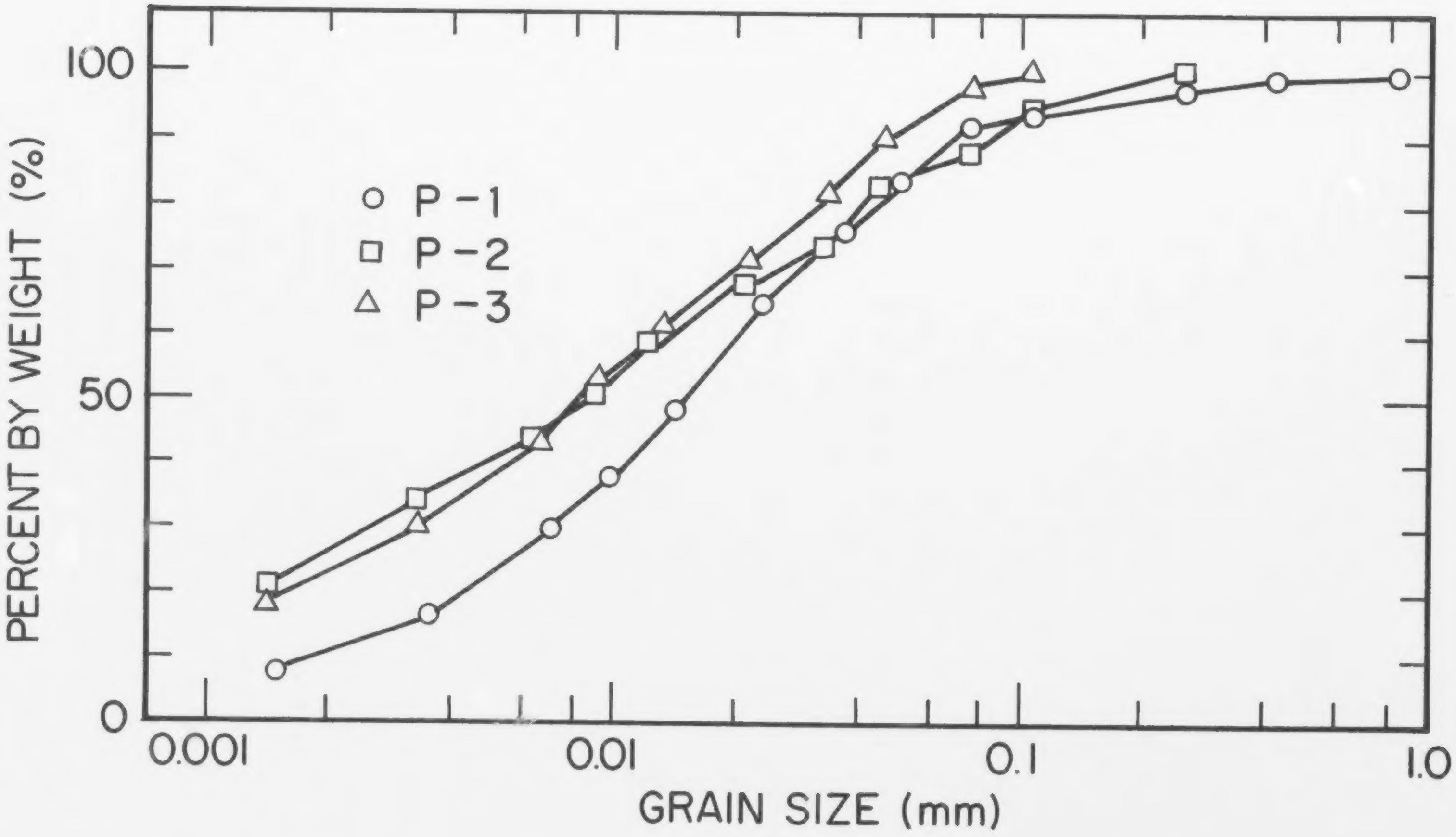

Figure 1b . Grain size accumulation curve (P-1, P-2, P-3).

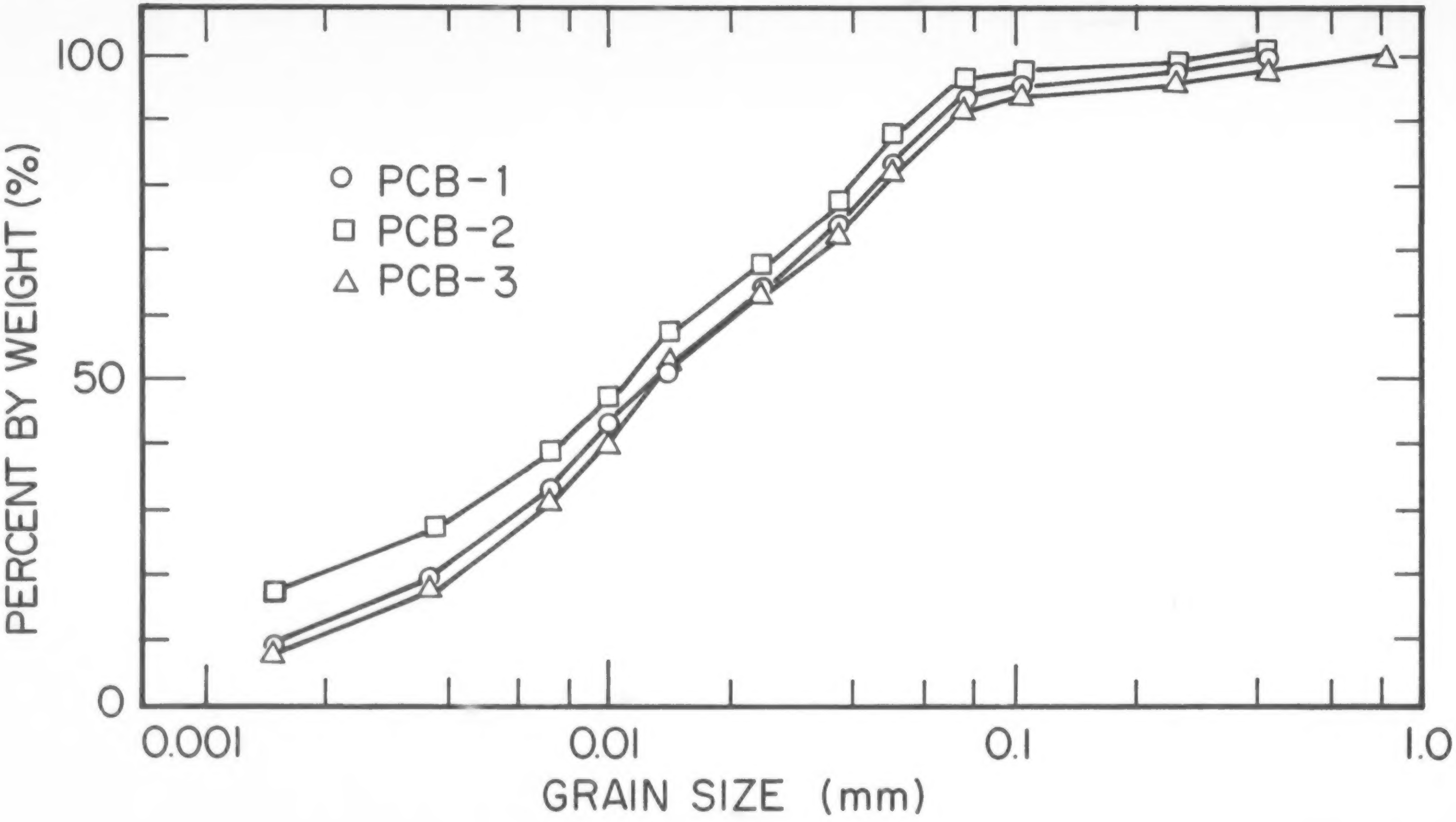

Figure 1a. Grain size accumulation curve (PCB-1, PCB-2, PCB-3).

TABLE 3. SAMPLE PROPERTIES

Sample	Content (ppm)		Ignition Loss (%)	True Specific Gravity
PCB-1	PCB	52.9	21.8	2.432
PCB-2		10.9	23.8	2.413
PCB-3		6.2	19.1	2.490
Hg-1	T-Hg[1]	43.3	22.0	2.289
Hg-2		11.3	13.4	2.438
P-1	PO_4-P	1.363	9.3	2.594
P-2		1.238	4.6	2.659
P-3		0.986	8.3	2.648
N	T-N	6640	26.7	2.234

[1] T-Hg, PO_4-P, and T-N mean total mercury, phosphate-P and total nitrogen, respectively.

$$C = 0.18 \cdot \frac{\Delta H}{J} \frac{1}{S}$$

where:

H = the mean range
J = the solubility rate, and
S = the safety factor.

For example, at Nagoya Port H, J, and S are 1.53 m, 0.0077, and 50, respectively, so C = 3.5 ppm. Consequently, sediments at locations Hg-1 and Hg-2 should be removed.

It is interesting to understand how the material is contained in the bottom sediments. An attempt was made to obtain the relationships between the contents per weight of dried sediment and particle size. Samples, except for PCB, were wet-sieved into several groups to prevent variation in the contents. The results are shown in Figures 2-4. The curve at Station Hg-1 has a peak of Hg concentration in the 105 μm to 250 μm range. In general, the contents of both PCB and mercury increase with decreasing particle size, but for nitrogen the content appears to increase with increasing particle size. It is usually considered that the amount adsorbed per weight of solid increases with the solid's specific surface area, which usually depends on decreasing particle volume. The high ignition loss led us to attempt to relate ignition losses to particle sizes. The relationship between ignition loss and nitrogen content is shown in Figure 5. Figure 6 shows ignition loss vs mercury content. The total nitrogen concentration varies linearly with ignition loss, but there is no correlation with mercury.

Sand used for the filtration test meets the standards set by the Japan Waterworks Association. The sand contains very little dust, clay or flat or weak grains. Table 4 shows the properties of the sand and Figure 7 shows the

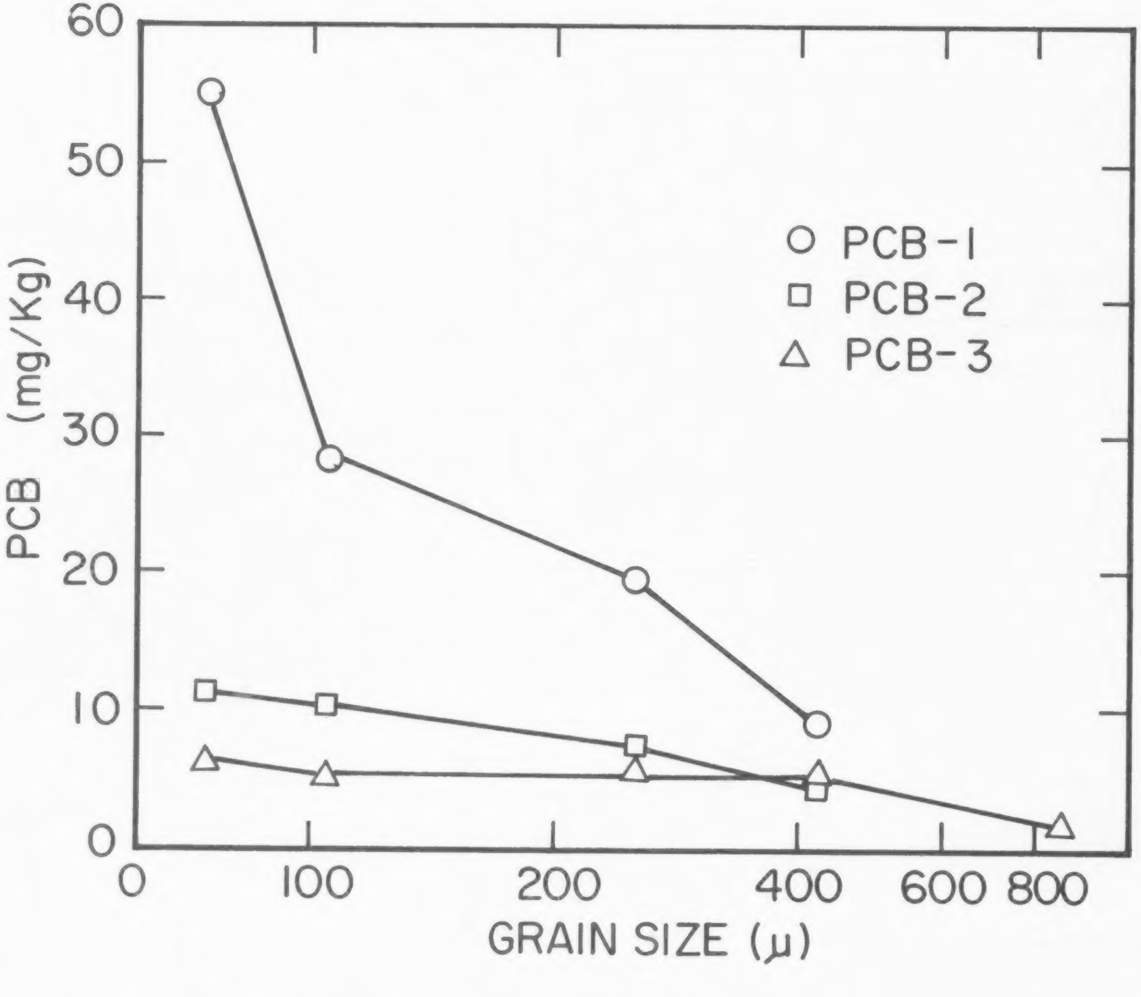

Figure 2a. Variation of concentration of PCB with particle size.

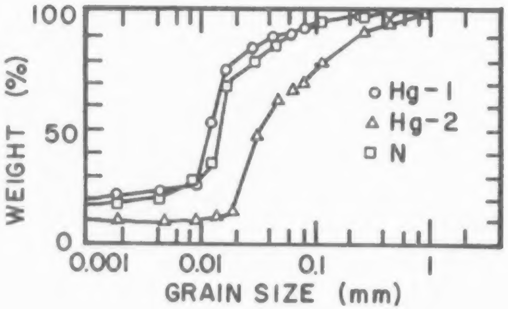

Figure 2b. Grain size accumulation curve (Hg-1, Hg-2, N).

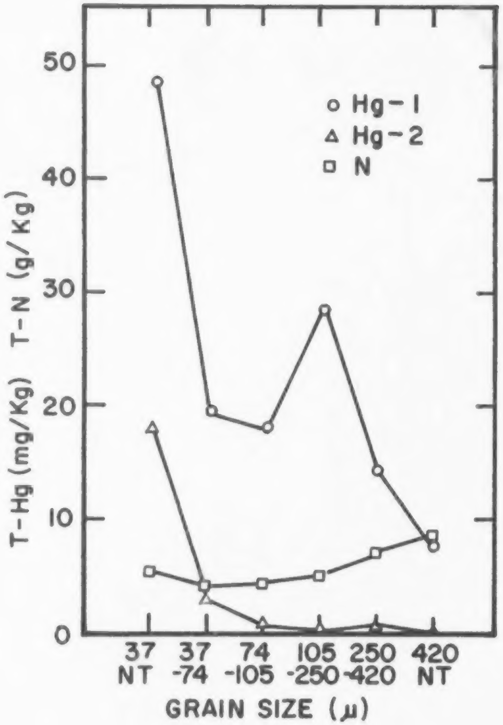

Figure 3. Variation of concentration of mercury and nitrogen with particle size.

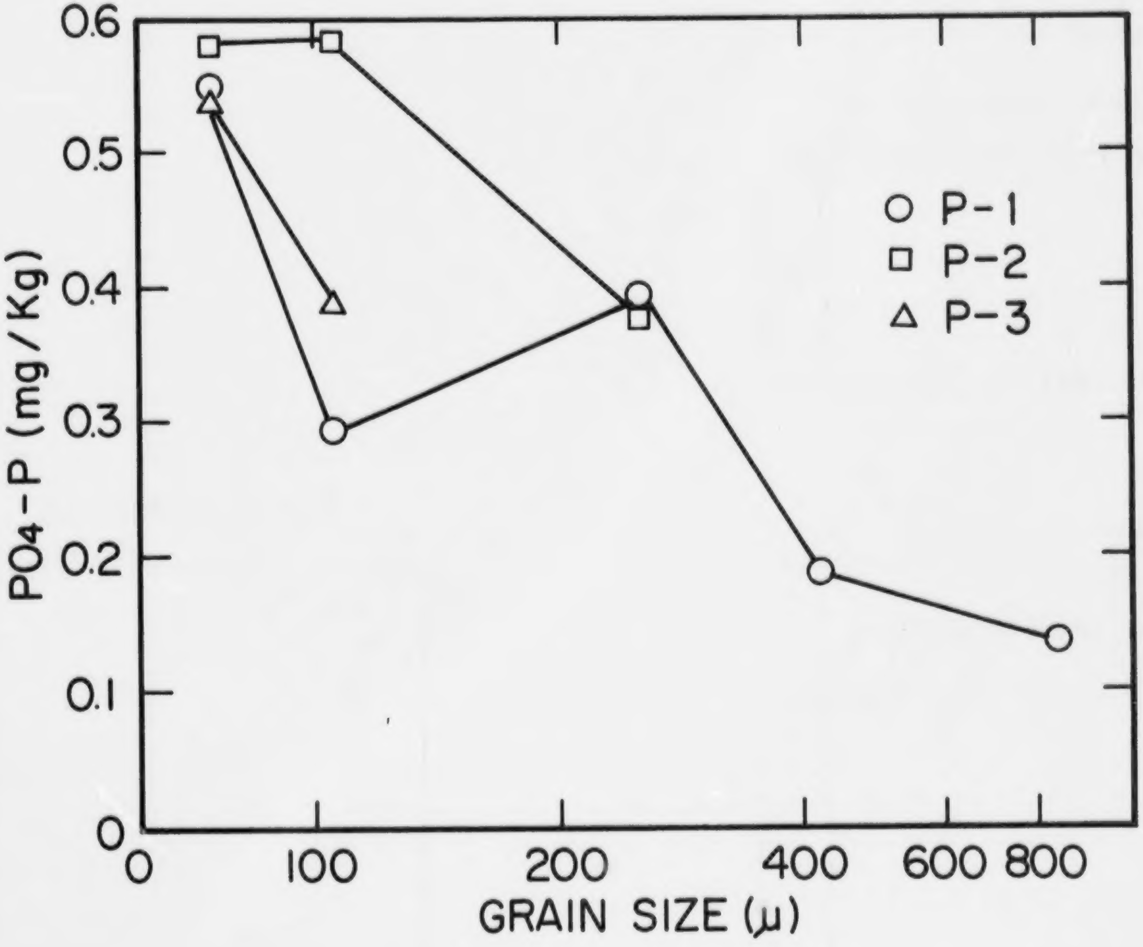

Figure 4. Variation of concentration of phosphate-P with particle size.

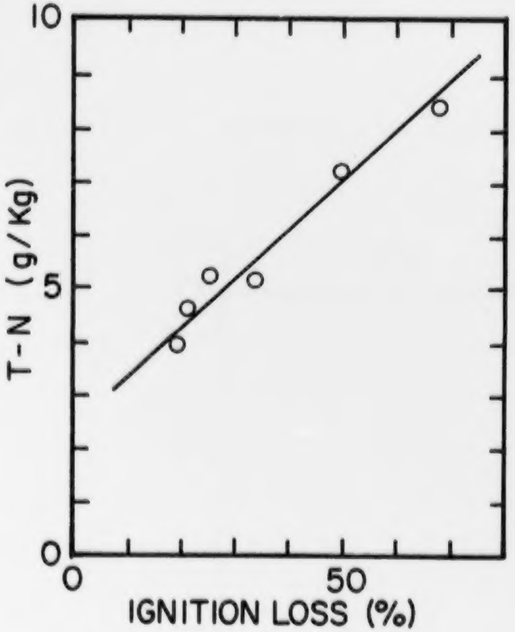

Figure 5. Variation of total nitrogen concentration with ignition loss.

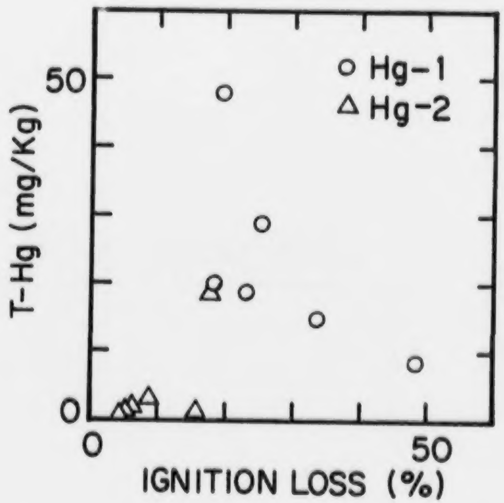

Figure 6. Variation of total mercury concentration with ignition loss.

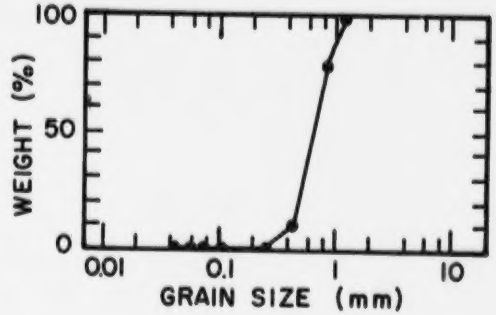

Figure 7. Grain size accumulation curve of filtration sand.

grain size accumulation curve. The coefficient of permeability of the sand layer as used for filtration is based on the constant-head permeation method. This value may differ from the permeability under use conditions.

TABLE 4. PROPERTIES OF FILTRATION SAND

Ignition Loss (%)	Effective Grain Size (mm)	True Specific Gravity	Porosity (%)	Coefficient of Permeability (cm/s)
0.545	0.405	2.610	41.6	0.154

PRINCIPAL ANALYSES

SEDIMENTATION ANALYSIS

Samples were sieved to below 420 μm under wet conditions. Suspensions were made by mixing the samples with 50 liters of either Tokyo Bay seawater or artificial seawater, as mentioned previously. The suspensions, at a predetermined concentration, were stirred at 40-60 rpm for 10 minutes in a settling cylinder 291 mm in diameter and 900 mm in height. Settling time was measured from the moment stirring stopped. The central part of the water column was sampled at fixed time intervals.

Increased settling rates caused by coagulants were examined under the conditions shown in Table 5. Time-dependent variations in SS vs material concentrations are given in Figures 8-11.

TABLE 5. SPECIFICATIONS FOR SEDIMENTATION ANALYSIS

Sample	SS (ppm)	Coagulant and Its Addition Concentration		Sampling Time (min)	Particle Size
PCB-1 PCB-2	3,000 6,000	no additives aluminum sulfate ferrous sulfate unslaked lime cation polymer anion polymer	 100 100 1,000 1 1	0, 5, 10, 15 30, 60, 120, 300, 600	sieved below 420 μm
Hg-1 Hg-2	50 400 3,000	no additives		0, 5, 10, 15 30, 60, 300 600, 1440	same as above
N	50 400 400 400	no additives no additives PAC cation polymer	 100 1	same as above	same as above

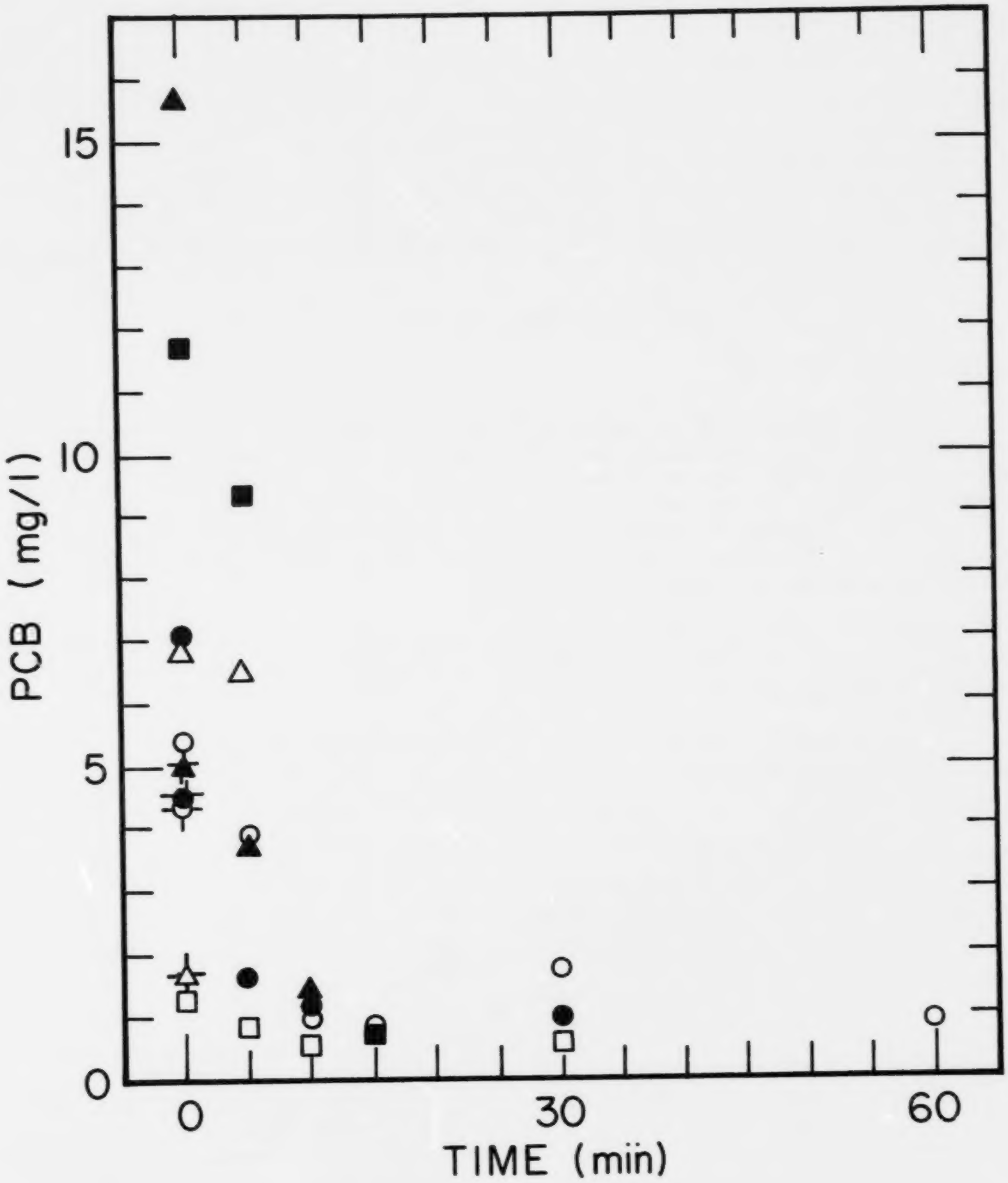

Figure 8. Variation of PCB concentration with time (see Table 8 for symbol identification).

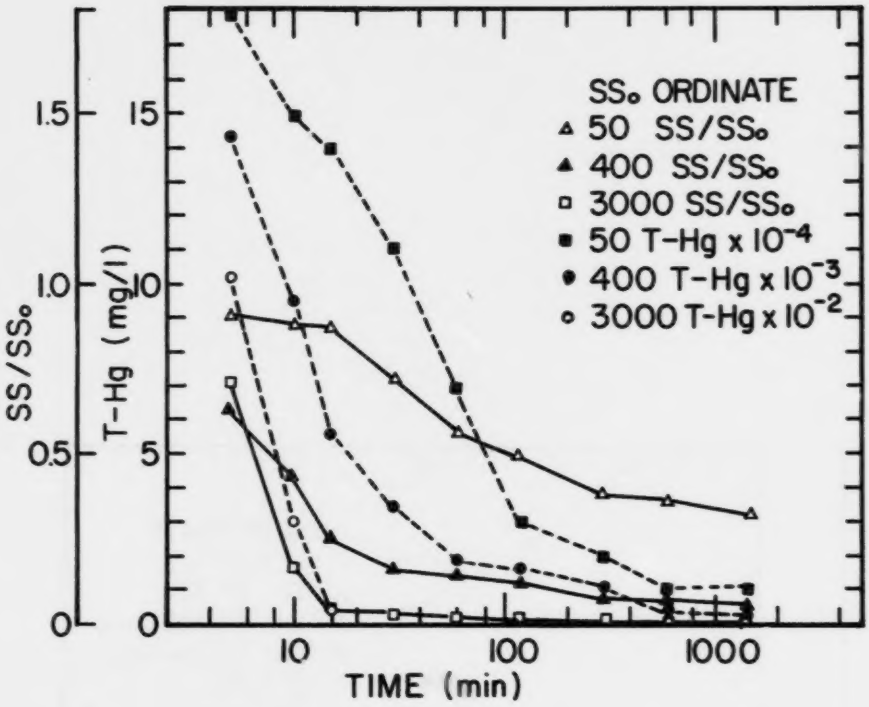

Figure 9. Variation of T-Hg concentration and SS with time (Hg-1).

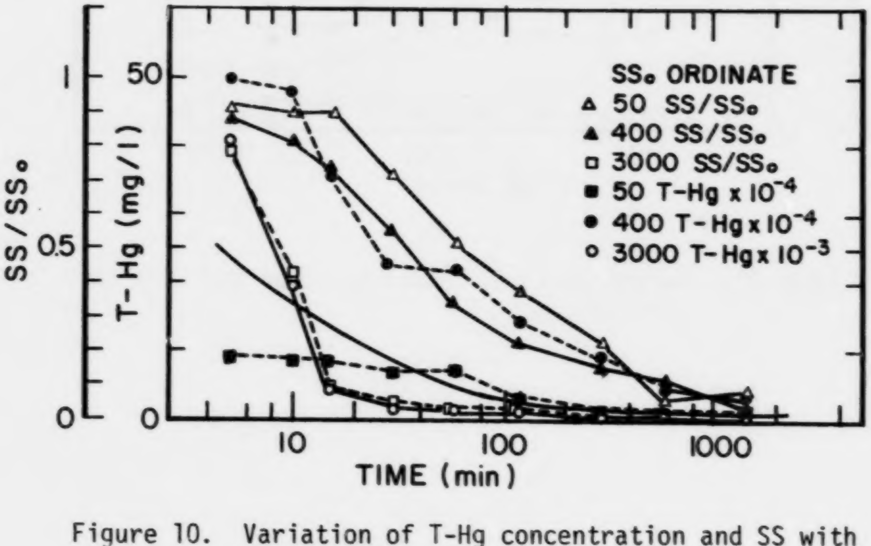

Figure 10. Variation of T-Hg concentration and SS with time (Hg-2).

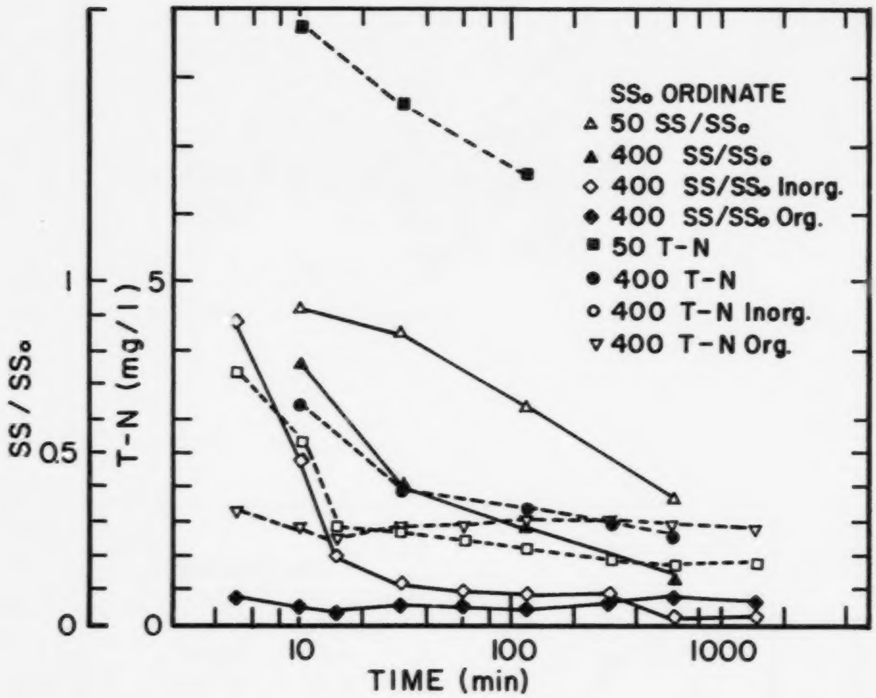

Figure 11. Variation of T-N concentration and SS with time (N).

Effluent standards require reclamation if the concentration is greater than 0.003 mg/l for PCB and 0.005 mg/l for total mercury. As in Figures 8-11 the relationships between SS vs material concentration are plotted in Figures 12-16, but the solid lines in these figures are based on the assumption that the material is part of the suspended solids, that is, the lines represent (SS) x (material concentration). In Figure 12 the measured values of PCB concentration deviate from the line, but Murakami's (1921) measurements show good agreement with the line. PCB has extraordinarily high chemical stability and is flammable only at very high temperatures. Generally speaking, PCB is not soluble in water, but the forced stirring makes it possible for PCB to dissolve to concentrations of 0.3 to 5 ppm. Also, studies have shown that if surface active agents coexist with the PCB, then its solubility increases. For example, 10 ppm of "Tween 80" in water is responsible for producing 10-20 ppm solutions of KC-200, KC-300, and KC-400 PCB (Isono and Fujiwara). Surface active agents are thought to be mixed in natural water in large quantities. Considering this possibility—that the water of Tokyo Bay contains surface active agents— the PCB contained in soil particles may dissolve in sea water from both the stirring and surface active agents and then adhere to particulates as they settle, consequently, reducing the water column concentrations.

To test the above assumption the following experiment was conducted. A mixture of artificial seawater, surface active agent, and PCB was allowed to settle and a sample of the upper water layer was taken after a two-hour settling period. The test conditions were:

SUBSTANCE	QUANTITY
PCB (KC-400).	3 grams
Surface active agent [Dodecyle-benzol sulfonate (DBS)]. . .	0, 0.03, 0.1, 0.3, 1.0, 10 mg/l
Seawater.	3 liters

The results of this experiment indicate that the surface active agent has no apparent effect on the solubility of PCB in water (Table 6).

TABLE 6. PCB SOLUBILITY AS A FUNCTION OF SURFACE ACTIVE AGENT CONCENTRATION

DBS Content (mg/l)	0	0.03	0.1	0.3	1.0	10.0
PCB solubility (mg/l)	0.0153	0.0155	0.0159	0.0158	0.0143	0.0155

The solid line in Figure 13 indicates good correlation between T-Hg concentration and SS when the latter exceeds 50 mg/l. It shows further, however, that the correlation decreases sharply when SS concentrations are less than 50 mg/l. For the Hg-2 sample (Figure 14), which is a mixture of two bottom sediments at Tokyo Bay and Nagoya Port, the solid line corresponds well with actual measurements. For P-1, P-2, and P-3 (Figure 15), there exists no

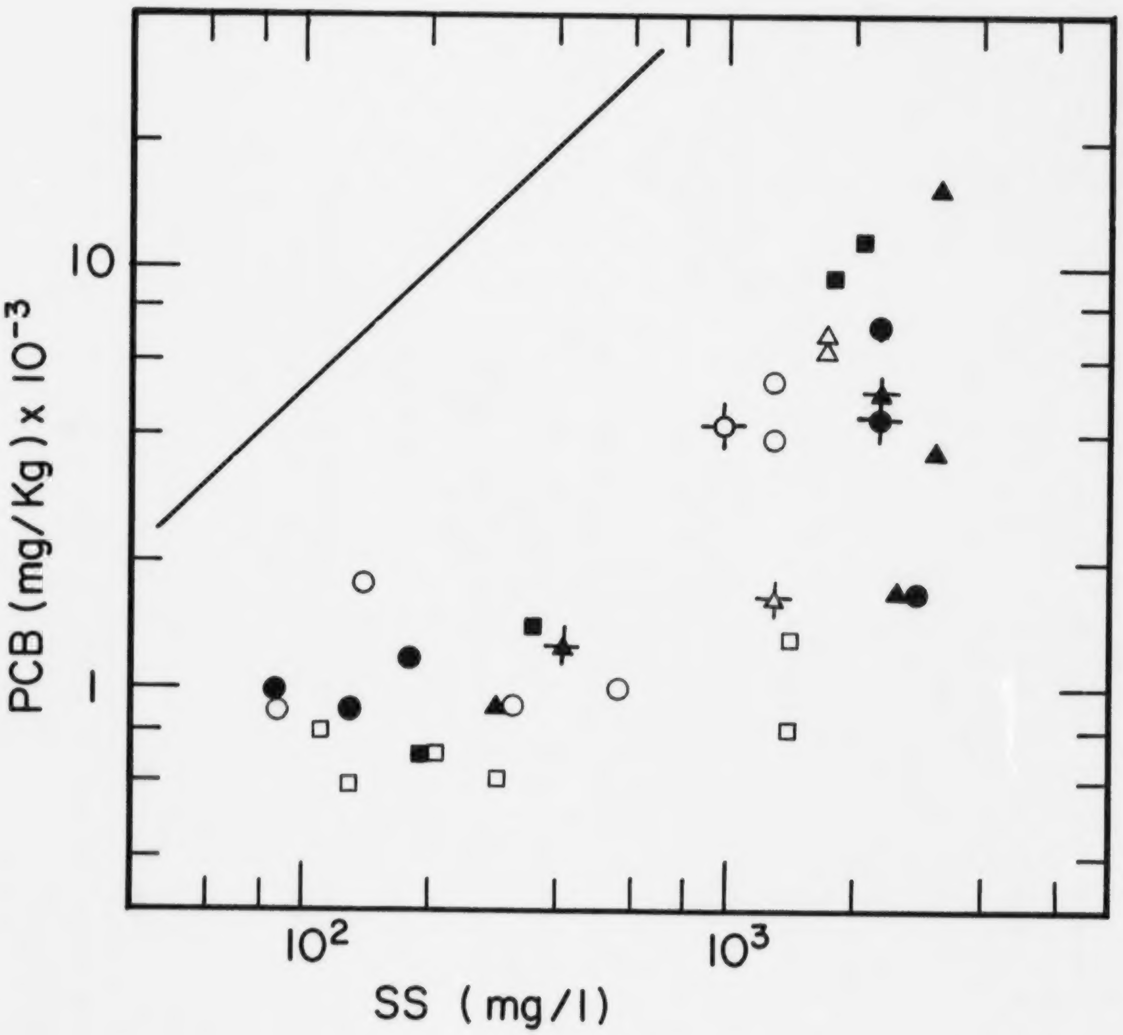

Figure 12. Variation of PCB concentration with SS (see Table 8 for symbols).

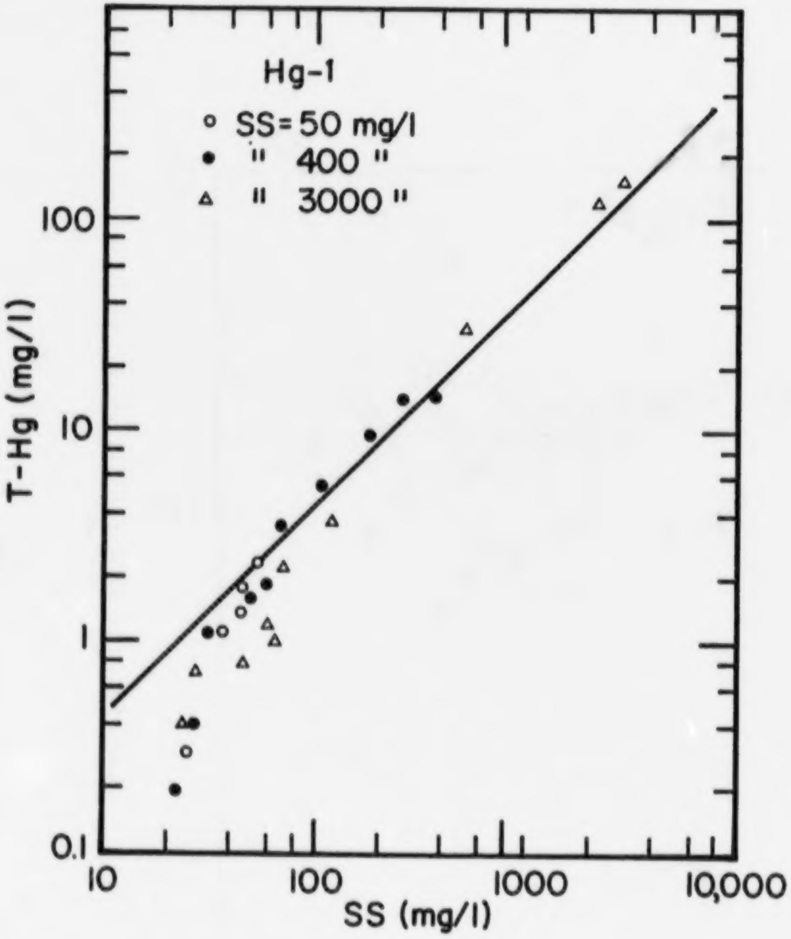

Figure 13. Variation of T-Hg concentration with SS (Hg-1).

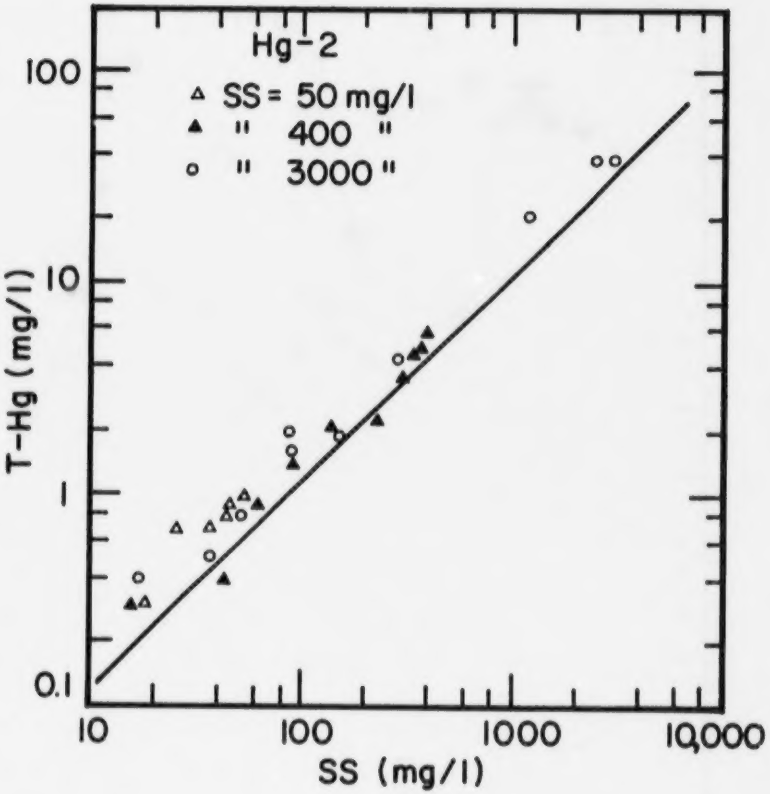

Figure 14. Variation of T-Hg concentration with SS (Hg-2).

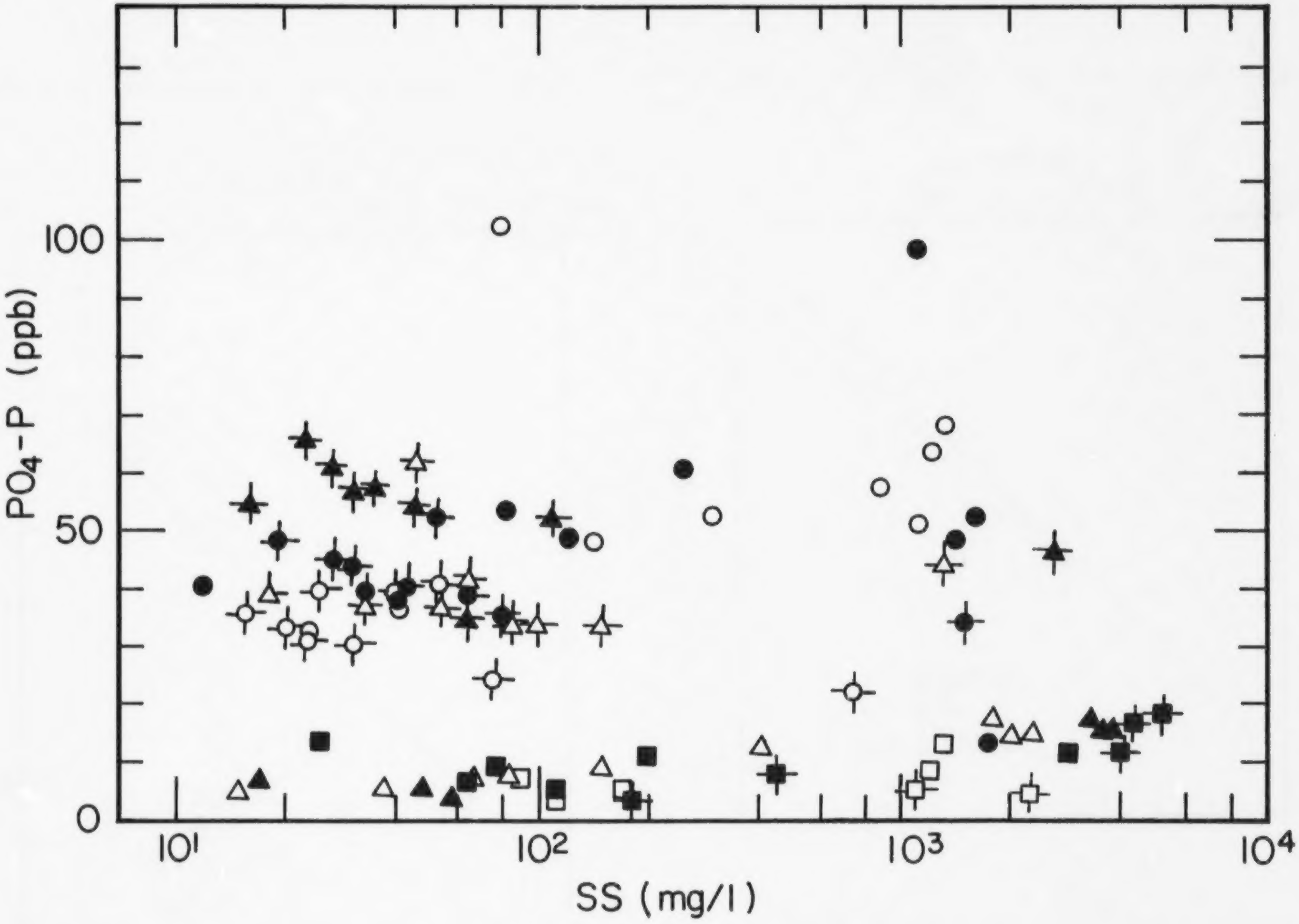

Figure 15. Variation of phosphate-P concentration with SS (see Table 8 for symbol identification).

relationship between SS and phosphate concentration, but it is deduced from the figure that coagulation agents such as aluminum sulfate, ferrous sulfate, and unslaked lime have the same effects as removal of soil particles. It is found that polymers have no effect. Phosphate-P dissolves in water from soil particles in the form of ions and exists in the suspension regardless of soil particles, but the inorganic coagulants and the ions unite and settle as solid particles. With regard to N the relationship between total nitrogen and SS is not strong, in addition none of the coagulants appear to remove nitrogen.

From the above results, it is clear that it is possible to monitor water quality by means of SS instead of direct measurements for total mercury or PCB, i.e., water quality may meet effluent standards by keeping SS less than 100 mg/l for Hg-1 and less than 300 mg/l for Hg-2.

Sedimentation of soil particles can be generally classified into three types: (1) discrete settling which occurs when SS is very low; (2) flocculant settling, and (3) zone settling. Consider a standard settling model based on the assumption that individual particles behave according to Stokes' Law. This model fits the type 2 settling classification. The following discussion shows how the settling curve takes the shape of that of the Hg-1 sample.

Assuming that the grain size accumulation curve takes the shape of a logarithmic normal distribution, and denoting the particle diameter by x, the curve can be described by the equation

$$F(x) = \frac{1}{\sqrt{2\pi\sigma}} \int_{-\infty}^{\ln x} \exp\left[- \frac{(\xi - m)^2}{2\sigma^2} \right] d\xi \qquad (1)$$

where F(x) denotes percent by weight of soil particles below x, $\xi = \ln x$, $m = \ln \sqrt{x_{84}\ x_{16}}$, $\sigma = \ln \sqrt{x_{84}/x_{16}}$. For the Hg-2 sample, from Figure 1, $m = 2.965$ and $\sigma = 0.999$.

According to Stokes' law, the settling velocity of particles having diameter x is represented by the equation

$$v = \frac{g(\rho_s - \rho_f)^z}{18\mu} x^2 \qquad (2)$$

Let p(x)dx represent the percent weight of soil particles having velocity v in the suspension whose initial SS is equal to SS_0. Let SS at a distance z from the surface be SS, and elapsed time after stirring be t, the equation

$$SS/SS_0 = \int_{-\infty}^{\sigma\sqrt{v}} p(x)\, dx \qquad (3)$$

is obtained. Since dF(x) = p(x)dx, v = z/t, it becomes

$$SS/SS_0 = \frac{1}{\sqrt{2\pi\sigma}} \int_{-\infty}^{\ln \sigma\sqrt{z/t}} \exp\left[- \frac{(\xi - m)^2}{2\sigma^2} \right] d\xi \qquad (4)$$

where $\sigma = \sqrt{18\mu/g\ (\rho_s - \rho_f)}$. The curve obtained from equation (4) is plotted in Figure 10. This ideal settling curve is influenced by convection due to a temperature differential between the upper and lower part of the water column and the diameter of the container. It is obvious from Figure 10 that sedimentation is better when SS_0 is high. Where SS_0 was 3,000 mg/l, SS was reduced to 1/10 of SS_0 in 30 minutes. By comparison the 50 and 400 mg/l initial suspensions show a slower sedimentation time than that of the theoretical model.

Increasing the sedimentation rate by using coagulants was investigated using a suspension of 400 mg/l of SS made from sample N and artificial seawater. PAC and cation polymer were added separately to the suspension in the amounts shown in Table 5. Figure 11 shows the test results. Sedimentation is enhanced by addition of an inorganic coagulant agent and even more by the organic coagulants.

The above results show that PCB and mercury bond strongly with soil particles. Therefore, to remove PCB or mercury from seawater, it is sufficient to remove the suspended solids. Some coagulants are effective in removing phosphate-P from the suspension, but they are not useful for removing nitrogen.

FILTRATION TEST

For this test samples were passed through a sand filter. The apparatus was a vertical cylinder 3.2 m high with a 0.291 m diameter. The filter column was packed with sand having grain size characteristics as shown in Figure 7.

The tests were run by pouring the suspensions into the cylinder and, with the valve left open at the bottom, periodically sampling the filtered water and analyzing it for mercury, PCB, and SS. During the experiment the water was maintained at a constant level by continually adding more of the suspension being tested. For example, Hg-1 measurements were made of the filtration ratio (which is defined by the flow rate of the filtered water) and the level of total mercury in the sand layer at the end of the test.

It is obvious that the majority of SS can be removed by filtration, therefore PCB and mercury should not penetrate the sand layer. But note that sediments deposited on the sand layer also act as a filter. These sediments were several millimeters thick.

Figure 17 shows variations in filtration rate with time. The filtration rate decreases sharply after several hours. The influences of water pressure over the sand surface on the filtration rate is also illustrated. A pressure difference of 1 m of depth slightly increases the filtration rate. Figure 17 also compares the filtration characteristics of the suspensions prepared from sea bottom sediments sieved blow 420 μm and 74 μm. The filtration rate of the suspension below 74 μm is greater than that of the suspension below 420 μm.

Total mercury in filtered water is shown in Figure 18. The test conditions under which the results in Figure 18 were obtained were the only ones that yielded detectable quantities of total mercury. In this test, concentra-

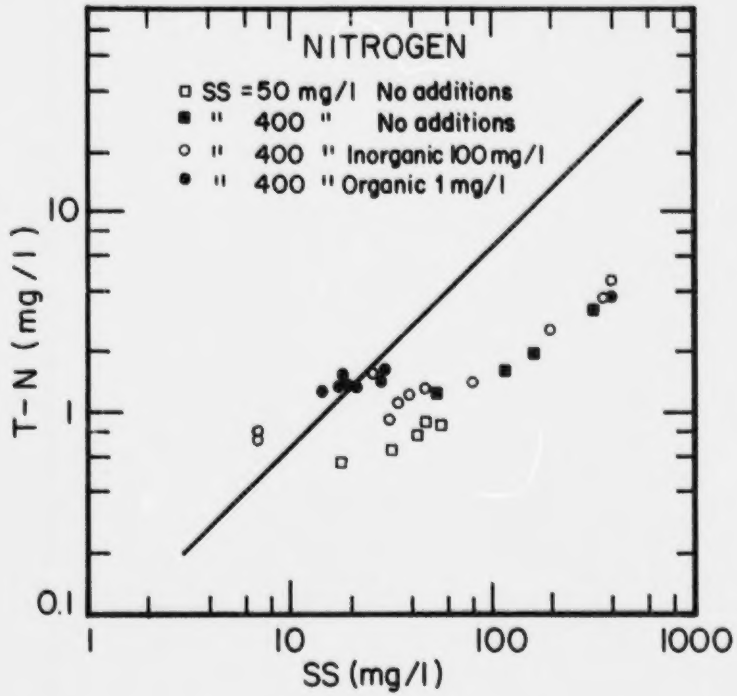

Figure 16. Variation of T-N concentration with SS.

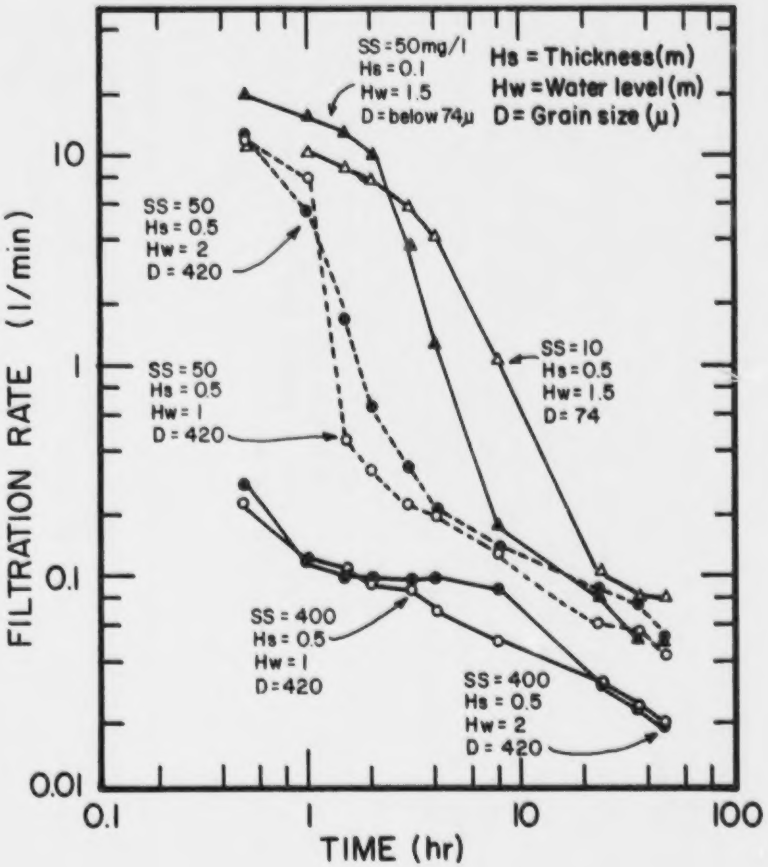

Figure 17. Variation in filtration rate with time.

tion decreased sharply with time. Even after only one hour the concentration of total mercury showed only 0.0012 mg/l, which easily meets the effluent standard of 0.005 mg/l.

The total mercury concentration at different depths in the sand layer was measured and the results are presented in Figures 19 and 20. In Figure 19 the mercury does not reach the bottom of the sand layer; in Figure 20 larger amounts of soil particles are found at the lowest part of the layer. But even in this case, total mercury is not detected in the filtered water. This leads to the conclusion that in the early filtration process almost all the voids of the sand layer are clogged by soil particles.

The significant conclusion from the above results is that, except for the early stages of reclamation work, a bulkhead built of earth and sand instead of expensive watertight structures may prevent PCB or mercury from re-entering the environment, i.e., filtration through a sand layer of proper grain size is an effective method for removing PCB and mercury.

SOLUBILIZATION TEST

The test was carried out according to the Methods for Investigating Bottom Sediments authorized by the Environmental Agency, except that artificial seawater and natural seawater were employed in preparing the mixtures of bottom sediments and water. The mixture was continuously stirred for four hours, and after being allowed to settle a bit, it was passed through filter paper. The filtered water was analyzed for contents. Conditions for the test are shown in Table 7.

TABLE 7. TEST CONDITIONS AND RESULTS

Sample	Mixing Ratio (g/ml)	Concentration After Filtering (mg/l)
PCB-1	50/100	0.0006
	25/100	ND
	3/100	ND
	0.3/100	ND
Hg-1	3/100	ND
	1.5/100	ND
	0.3/100	ND
	0.03/100	ND

Mixing ratio is the ratio of soil dry weights (g) to the volume of the mixture (ml).

ND means "not detectable," i.e., 0.0005 mg/l or less for PCB, and 0.0001 mg/l for total mercury.

The Methods for Investigating Bottom Sediments defines the solubilization ratio as:

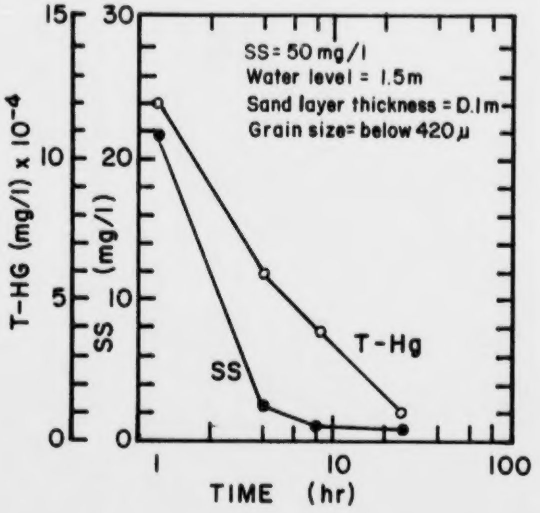

Figure 18. Variation of T-Hg concentration and SS of filtered water with time.

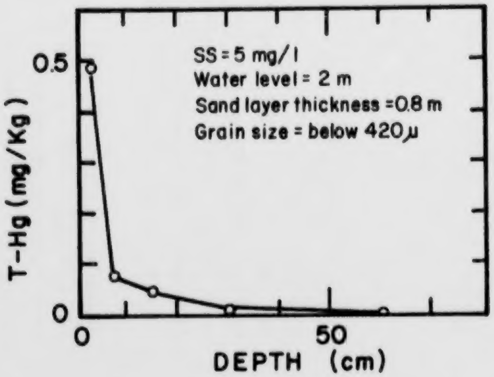

Figure 19. T-Hg distribution in sand layer.

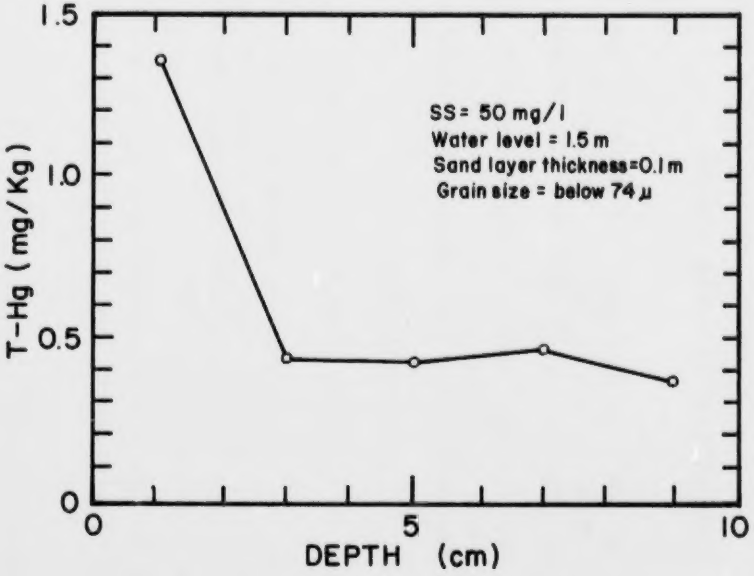

Figure 20. T-Hg distribution in sand layer.

$$\text{Solubilization ratio} = W_2/W_1$$

where W_1 is the amount of toxic matter (μg) contained in the bottom sediment sample, and W_2 is the amount of toxic matter (μg) which is contained in a volume of filtered water equivalent to the mixture prepared in the test.

According to this definition, the solubilization ratio is calculated to be less than 7.7×10^{-3} for Hg-1 and 5.57×10^{-5} for PCB-1. The preliminary criteria for sea bottom sediments were calculated using these values.

CONCLUSIONS

The main conclusions to be drawn from this study are:

1) The finer the particles of the bottom sediments, the higher the concentration of PCB and total mercury. They adhere strongly to soil particles and it is therefore important to remove fine suspended solids to improve wastewater quality at reclamation sites.

2) Because a sand layer can prevent suspended solids from penetrating, a bulkhead filter built with earth and sand is capable of removing PCB and mercury.

3) Phosphate-P is removed by addition of aluminum sulfate, ferrous sulfate, and unslaked lime. Except for the nitrogen in soil particles, nitrogen will not be removed by any coagulation agent.

4) Total mercury concentration in the sand layer is extremely high but decreases abruptly with depth, leading to the conclusion that almost all suspended solids remain in a thin layer in the upper part of the sand layer. The filtration rate decreases sharply with time.

5) A surface active agent did not dissolve PCB in water under the test conditions employed here. During the test PCB was observed to fall in spherical particles after stirring stopped.

NOTES

1. In the sedimentation test, the direction of filtration was in the vertical direction, i.e., the sand layer was horizontal, allowing suspended solids to fall on the surface. This test method is not capable of examining the permeation of suspensions through bulkheads built up with earth and sand. But for secondary treatment of wastewater that does not meet the effluent standards, filtering contaminated suspensions through a sand layer is an effective method of improving water quality.

2. Regarding the solubility of PCB in water: the surface active agent has no effect, but it is possible that the deviation between the solid line and the measurements in Figure 12 may be due to oil, because PCB is soluble in oil. We intend to clarify this point.

TABLE 8. SYMBOLS (APPLICABLE TO FIGURE 8, FIGURE 12, FIGURE 15)

coagulation agent SS$_0$ (ppm)	no addition	aluminum sulfate	ferrous sulfate	anion polymer	cation polymer	unslaked lime
3,000	○	△	□	-○-	-△-	-□-
6,000	●	▲	■	-●-	-▲-	-■-

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THE FILTERING EFFECT OF CONTAINMENT WALLS ON SUPERNATANT FROM CONTAMINATED DREDGE MATERIAL

Takeshi Monji
Port and Harbor Research Institute
Ministry of Transport, Japan

ABSTRACT

In designing revetments for disposal sites, care must be taken to prevent pollution of the nearby environment from permeation of contaminated substances through the containment wall. This paper describes the results of the following experiments:

1) Heavy metal adsorption on clay
2) Filtration effects of sand backfill
3) Testing of steel piling seals for permeability

These experiments produced useful data to help design containment walls for land reclaimed with dredged material.

INTRODUCTION

The large cities of Japan, such as Tokyo, Osaka, and Nagoya are located in the coastal area. The rapid economic development of the 1960s caused environmental pollution to become an important problem along the coast, especially on the bottoms of some harbors, rivers, lakes and marshes. Contaminated substances with undesirable effects on the surrounding environment accumulated in these areas. Sources of these sediments are usually from waste discharges from manufacturing plants, mines, urban sewage, or agricultural effluents.

To cope with this pollution, the government imposed legal restrictions on effluent discharges and began to dredge the polluted sediments. This policy resulted in gradual improvment of the environment, but large quantities of sediment and urban waste are a continuing problem. One of the most difficult tasks is to find disposal sites for this sediment and waste. In Japan there is no space for land disposal. Therefore, sediment and waste has usually been managed as reclaimed land in a port area. After a period of time, these reclaimed lands have been used effectively for port facilities, urban redevelopment, distribution stations, and other uses. However, it is becoming more difficult to find appropriate disposal sites. Construction of revetments for dredged material has become more complex because of the technical problems associated with building on thick deposits of soft soils and in deep water. The most important consideration in constructing a containment wall for this

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purpose is to prevent secondary pollution due to permeation of contaminated substances through the wall. To construct an absolutely waterproof revetment is not impossible, but it is expensive and unnecessary. The author and his colleagues have begun studies to determine a safe, economical structure for revetments. This paper is an interim report of these studies, particularly on the filtering effects of these porous containment walls.

CONTAINMENT WALL CHARACTERISTICS

TECHNICAL CONSTRUCTION PROBLEMS

Containment walls for land reclamation have different characteristics than conventional wharves and revetments. First, conditions at the construction site are difficult because of soft ground and deep water. Second, design variables such as quality of material for reclamation, the height of the structure, and the residual water level are more severe. Third, and most important, is to prevent contaminated substances from permeating the wall. Though we can cope with the first two problems using conventional techniques, the third problem has unknown factors such as sand filtration effects and permeation rates of contaminated substances.

In Japan two types of containment walls have usually been constructed—the gravity type (caisson, cellular, stone masonry, etc.) and the sheet pile type (double wall, steel pile, etc.). Figure 1 shows typical Japanese revetments for reclaimed land. Methods of protection against permeation of contaminated substances are discussed for both types.

TECHNIQUES TO PREVENT PERMEATION OF CONTAMINATED SUBSTANCES

Techniques to prevent permeation of contaminated substances through revetments are shown in Figure 2. They may be classified into 5 groups:

1) Protection Sheet. Many kinds are commercially available and are effective for prevention of pollution. However, more study of durability and sheet arrangement is necessary to make practical use of this technique.

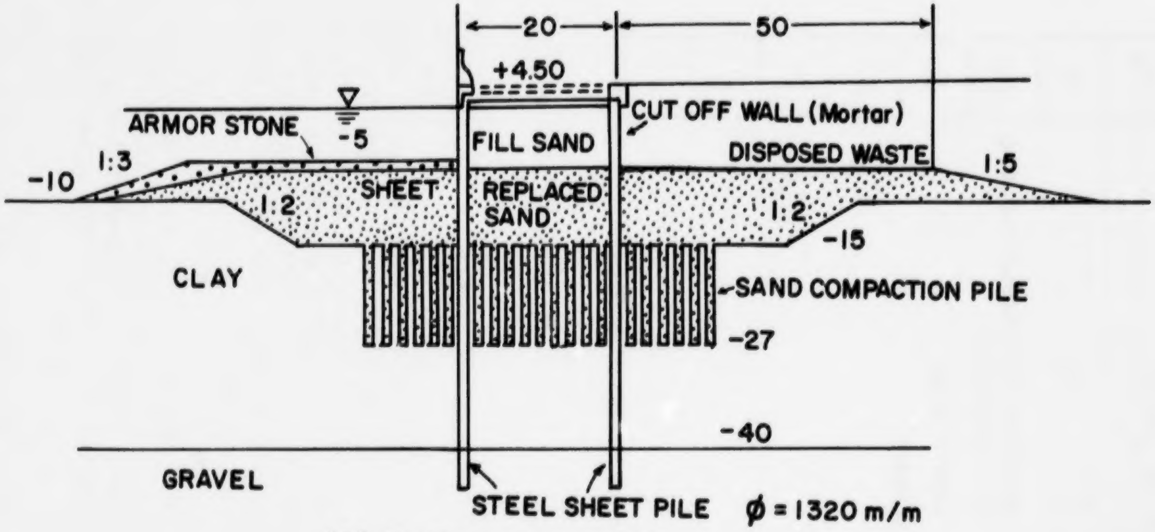
2) Backfill. This is often used as a protection method but a precise determination of the relationship of thickness and grain size distribution to permeation of pollutants still remains unsolved because the sand filter mechanism is not yet fully understood.

3) Sheet Pile. Piles are usually driven into the impermeable layers.

4) Grout. Sheet pile joints are sometimes sealed with asphalt, urethane, or mortar.

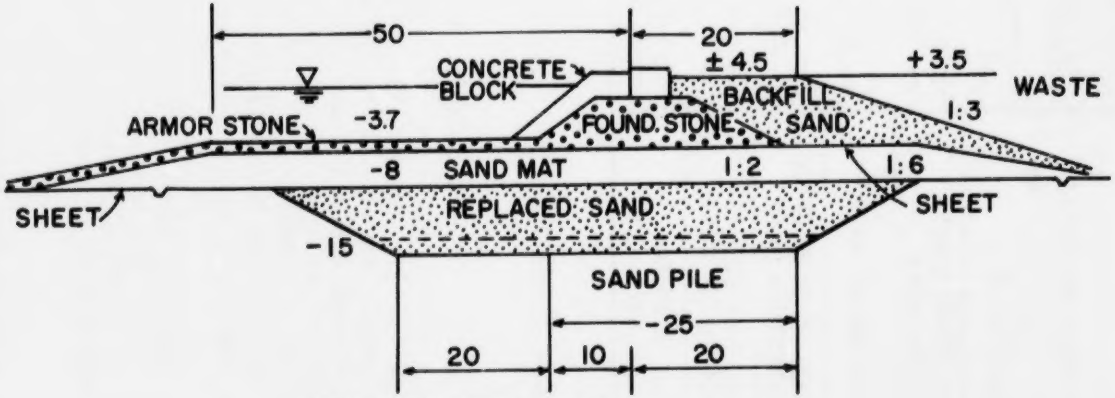
5) Direct Mixing. Direct mixing of asphalt or cement mortar with the disposed material. This causes soil stabilization.

In constructing revetments it is impractical to make them perfectly watertight. To prevent as much pollution as possible is the objective. Thus,



20
50
+4.50
ARMOR STONE
-5
CUT OFF WALL (Mortar)
FILL SAND
DISPOSED WASTE
-10
1:3
SHEET
REPLACED SAND
1:2
1:2
1:5
-15
CLAY
SAND COMPACTION PILE
-27
-40
GRAVEL
STEEL SHEET PILE
ϕ = 1320 m/m

(A) SHEET PILE TYPE



50
20
CONCRETE BLOCK
±4.5
+3.5
WASTE
BACKFILL SAND
1:3
ARMOR STONE
-3.7
FOUND. STONE
-8
SAND MAT
1:2
1:6
SHEET
SHEET
REPLACED SAND
-15
SAND PILE
-25
20
10
20

(B) GRAVITY TYPE

Figure 1. Typical containment walls for reclaimed land using dredged materials as fill.

the purpose of the experiments described in this paper is to determine the filtering effect of sand backfill and fill on the permeation of contaminated substances and, finally, to determine the proper design criteria for revetments.

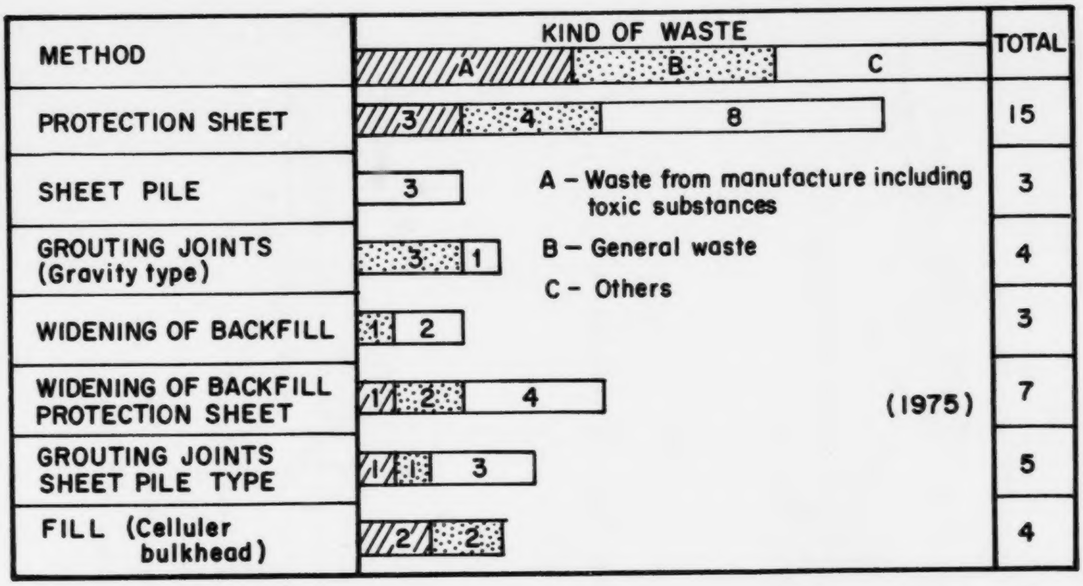


Figure 2. Techniques to prevent permeation of contaminated substances.

SAND BACKFILL AS A FILTER FOR HEAVY METALS

Clay mainly consists of silica, alumina and water. Some kinds of clay include magnesium, iron, alkaline-metal and alkaline-earth-metal. Clay grains are negatively-charged from ion exchange in the crystals of clay and from imperfections on the surface of clay particles. Though the electric charge depends on a combination of factors, heavy metal is generally adsorbed to clay. This is why bottom sediments in the coastal areas and rivers in Japan include some heavy metals which are deleterious to health.

In dredging these contaminated sediments and disposing of them as reclaimed land, the most important considerations are elution of toxic substances by dredging, disposal of the supernatant water of dredged sediment and the permeation of contaminated substances through the containment area revetment.

If clay is put into a solution containing heavy metal and agitated, the heavy metal is adsorbed to the clay. So when contaminated sediments are disposed of as reclaimed land, heavy metals in the water will be adsorbed to the suspended soils and these suspended soils will be filtered out by the backfill sand in the revetment.

By knowing the rate of adsorption of heavy metal to clay and the filtering capacity of sand backfill for suspended soil, the concentration of heavy metal in the effluent from contaminated substances can be controlled at a level which will satisfy the requirements.

These considerations allow revetments to be designed safely and economically by determining the kind and thickness of backfill sand. The filtering effect depends on both adsorption of heavy metal to clay minerals and the filtering out of suspended soils. The amount of heavy metal adsorbed to the clay varies with the kind of clay, type of heavy metal, concentration of seawater, and the pH of the suspension. The filtering effect of sand changes with the particle size of the sand, particle size of the clay, water pressure, density of sand and other factors.

CHARACTERISTICS OF MARINE CLAY AND BACKFILL SAND IN JAPANESE PORTS

Coastal Japan is covered with soft alluvial clays which come from rivers and in some cases derive from volcanic ash. Clay minerals are the result of chemical weathering of rock and consist mainly of kaolinite, montmorillonite and illite.

Clay particles are negatively charged. Cations such as heavy metal ions are thereby adsorbed to the surface of the clay particles. The higher the electric charge or the smaller the radius of the cations, the stronger the adsorptive adherence. The present experiments do not investigate in detail the chemical reaction of clays. At this time general information is more useful in designing containment walls.

Soil samples, including sea bottom clay or sand used for backfill were collected from the various locations shown in Figure 3. Using these samples, several series of experiments were made on heavy metal adsorption to clay and on the filtering effect of backfill sand. The physical and chemical characteristics of the clay and sand are given in Figures 4 and 5 and Table 1. With few exceptions all the clays used have similar physical properties.

The sands have a similar pattern of grain size distribution, somewhere between the Takahagi sand and the Toyoura sand used in the author's laboratory. Backfill sand samples were made by mixing these two sands for the experiments.

EXPERIMENTS ON CLAY ADSORPTION

FACTORS CONTROLLING THE ADSORPTION OF HEAVY METAL

The purpose of this experiment was to determine the rate of heavy metal adsorption on to clay. Lead, cadmium and mercury were the heavy metals used in this experiment and Kawasaki clay was used for the suspension. The main factors considered in the experiments were the concentration of the heavy metals and suspensions in solution. Additional factors investigated were agitation time, mesh size of filter, position of the agitating apparatus and pH.

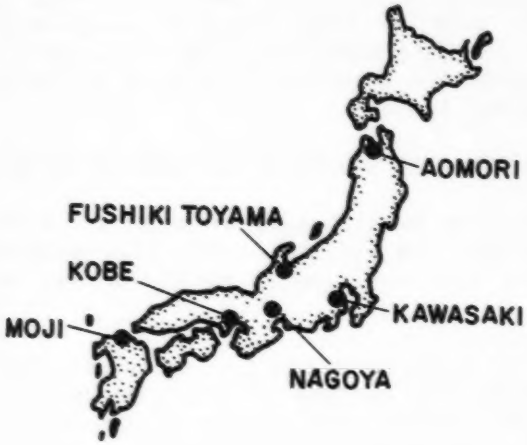


Figure 3. Main ports in Japan (clay and backfill sand from these ports were used in the experiment).

The heavy metal solution was prepared by dissolving bichloride of mercury, cadmium nitrate and lead nitrate in fresh water and ion-exchanged water. Five liters of undiluted solution, including clay, were agitated in a cylinder by rotary action for 30 minutes. After filtering, the concentration of heavy metal, turbidity, pH and other factors were measured.

Results of this experiment are:

$$A = \frac{Co - C}{C_o} \times 100$$

A: Adsorption percentage
Co: Concentration of solution with heavy metal
C: Concentration of solution after being adsorbed onto clay

(1) Adsorption of mercury to clay increases in proportion to the logarithm of turbidity, i.e. 70% for 100 ppm of turbidity and more than 80% for 500 ppm of turbidity. Lead and cadmium show the same adsorptive tendency as mercury (see Figure 6).

TABLE 1. CHARACTERISTICS OF CLAYS IN JAPAN

Characteristics		Aomori	Kawasaki	Nagoya	Fushiki Toyama	Kobe	Moji
		Clay Type					
Specific gravity (Gs)	--	2.56	2.69	2.68	2.66	2.70	2.70
Liquid limit (Wl)	%	70.4	75.6	64.5	63.8	100.4	76.5
Plastic limit (Wp)	%	46.7	37.8	30.9	32.9	39.2	29.6
Plasticity index (Ip)	--	23.7	37.8	33.6	30.9	61.2	46.9
Gravel	%	0.5	0	5.1	0.2	0.1	0.3
Sand	%	14.0	4.7	17.1	19.5	1.1	10.2
Silt	%	55.2	38.3	34.3	51.9	36.3	30.5
Clay	%	30.3	57.0	43.5	28.4	62.5	59.0
Classification	--	F	F	F	F	F	F
Total-mercury	ppm	0.36	0.33	0.23	0.48	0.17	0.05
Cadmium	ppm	0.75	0.23	0.32	0.64	0.24	0.10
Lead	ppm	56.1	39.4	41.2	35.9	40.2	19.9
Sulfide	mg/g	1.90	0.28	0.22	2.30	0.21	0.06
pH	--	7.9	8.4	7.9	8.0	7.8	7.7
O.R.P.	100 mV	-3.10	-1.40	1.60	-3.40	0.60	1.40

(2) Adsorption of heavy metal to clay is low in acid solutions ($pH \geq 6$) and high in alkaline solution ($pH \leq 7$). This tendency is more noticeable for cadmium and lead (see Figure 7).

(3) Adsorption of each metal when all three metals are mixed together is the same as that for individual metals.

(4) Changing agitation time (0.5-60 minutes) has no influence on adsorption.

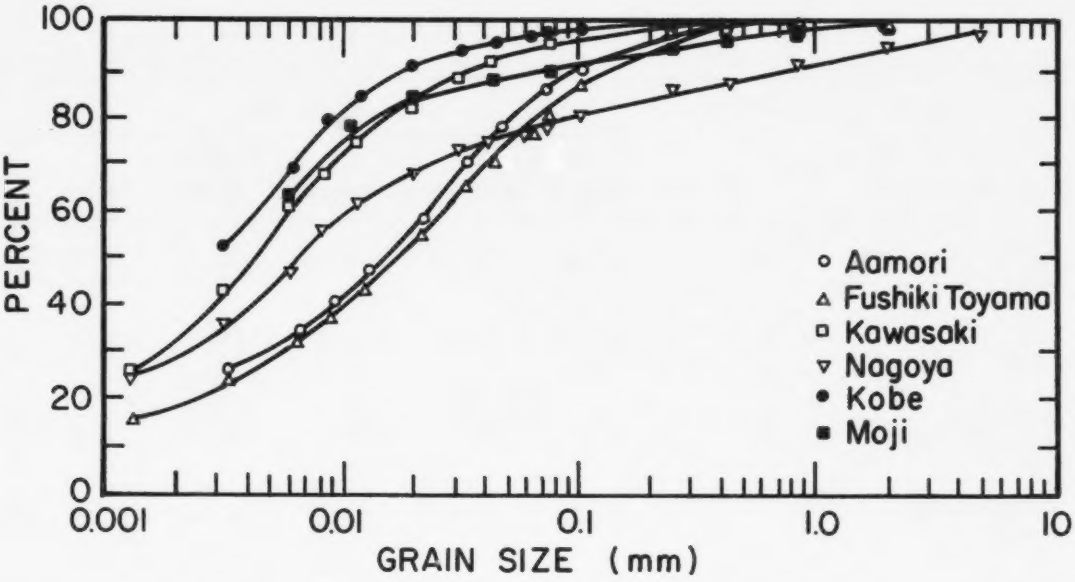


Figure 4. Grain size distribution of clay.

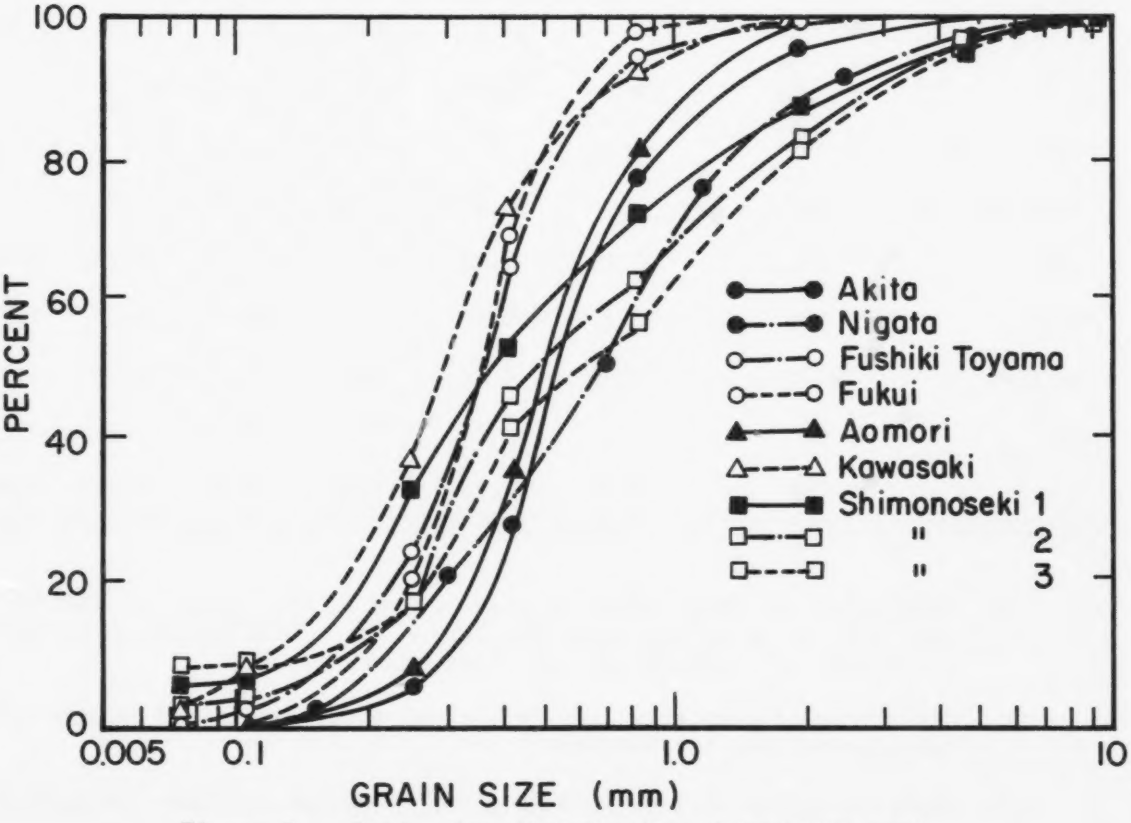


Figure 5a. Grain size distribution of backfill sand.

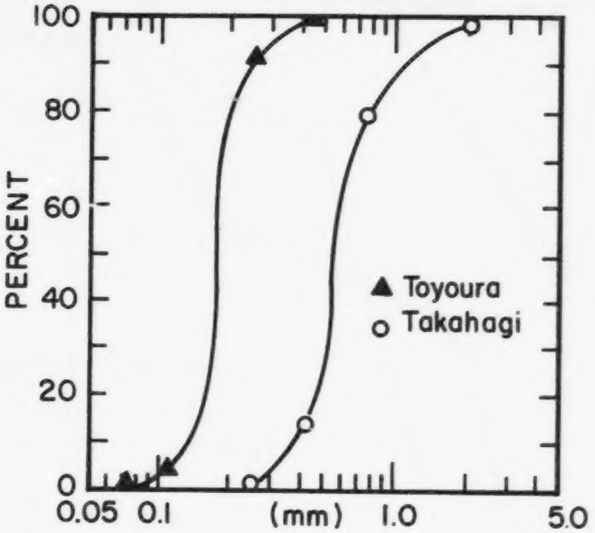


Figure 5b. Grain size distribution of sand used for the experiment.

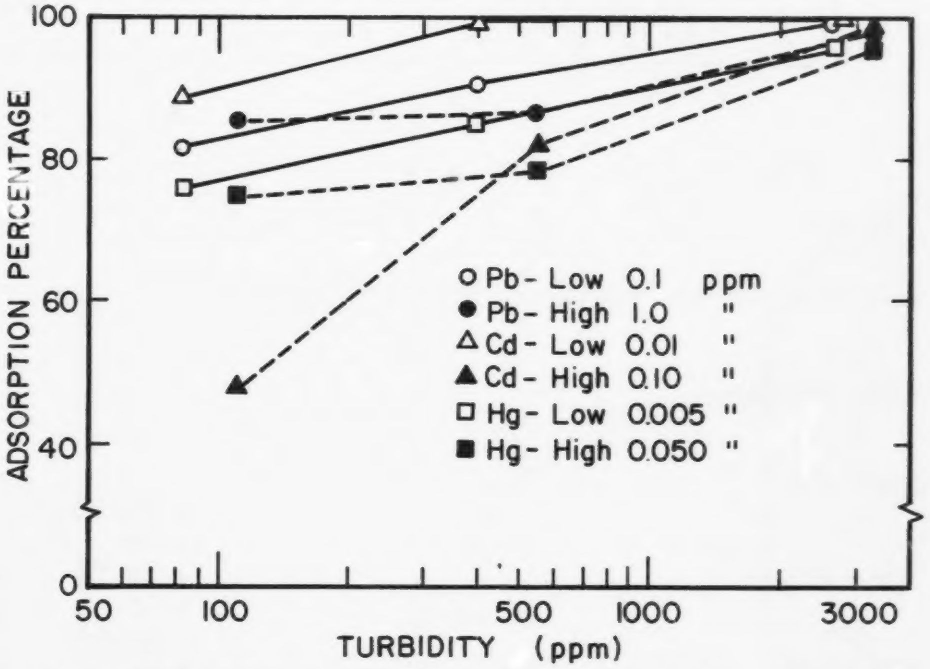


Figure 6. Adsorption of heavy metal onto clay. Note: Adsorption percentage A is defined as

$$A = \frac{Co - C}{C_o} \times 100$$

A: Adsorption percentage
Co: Concentration of solution with heavy metal
C: Concentration of solution after being adsorbed onto clay

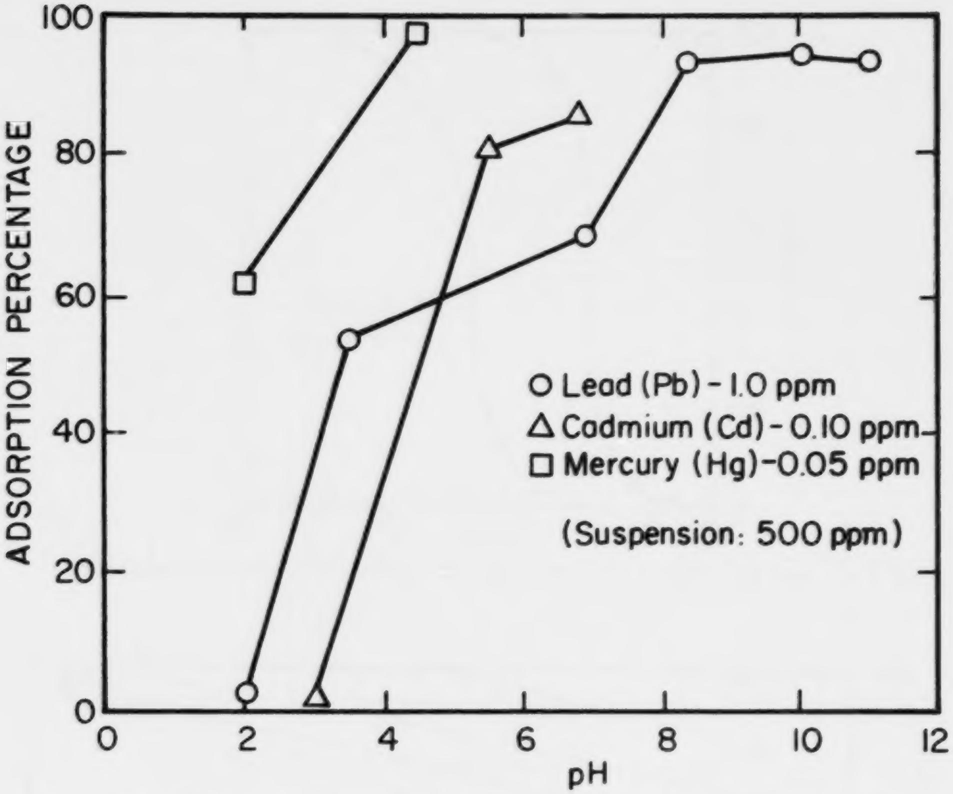


Figure 7. Relationship between pH and adsorption.

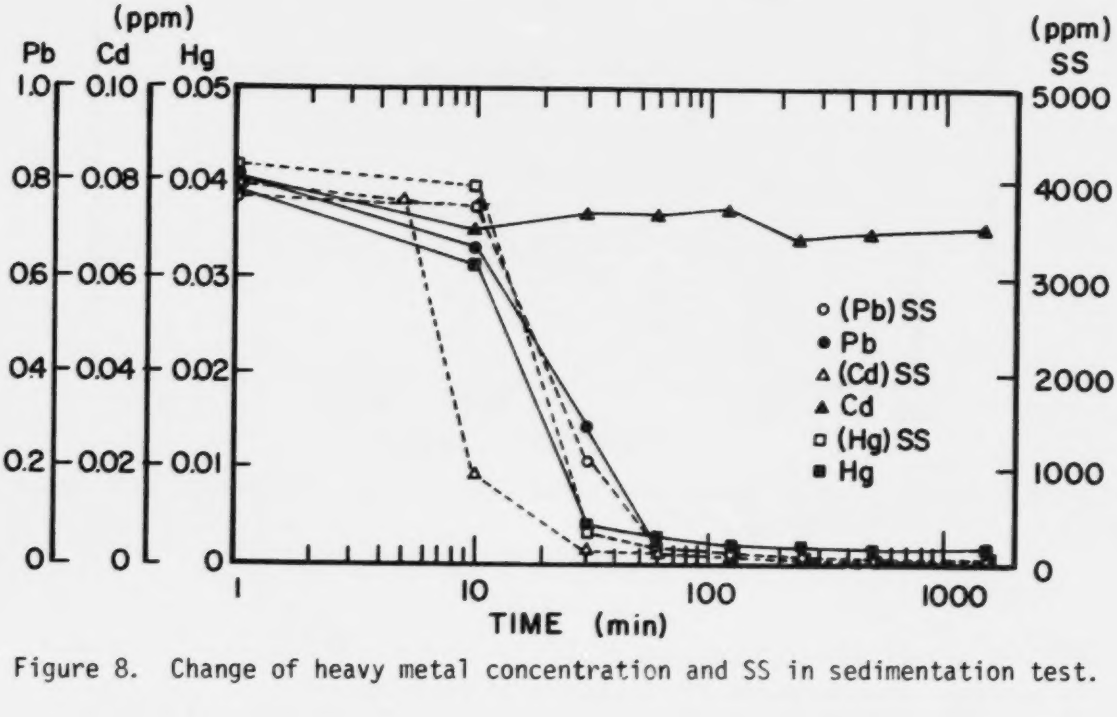


Figure 8. Change of heavy metal concentration and SS in sedimentation test.

EXPERIMENTS ON ADSORPTION AND SEDIMENTATION OF HEAVY METAL IN SEAWATER

The purpose of this experiment was to investigate heavy metal adsorption onto clay, and sedimentation of heavy metal in seawater. The authors used Kawasaki clay in suspension plus mercury, cadmium, and lead as heavy metals. The seawater was from Yokosuka in Tokyo Bay. The heavy metal solution was prepared by the same method as described above. The experimental apparatus was an acrylic cylinder (d = 190 mm, h = 100 cm). The prepared solution was poured into the cylinder to 90 cm in depth and allowed to settle after 1 minute of agitation. The solution was sampled by siphon at 45 cm of depth and analyzed. The results of this experiment are

(1) Clay particles flocculate in seawater and are deposited more rapidly than in freshwater.

(2) The concentration of mercury and lead in the solution decreases in relationship to the sedimentation of clay particles (SS), but the concentration of cadmium in the solution is unaffected by the sedimentation of clay particles (see Figure 8).

This may be because substances in seawater which are less easily ionized than cadmium are adsorbed onto the clay.

EXPERIMENTS ON ADSORPTION OF HEAVY METALS TO SANDS

The purpose of this experiment was to investigate heavy metal adsorption onto clay which is mixed with sand. Factors such as clay fractions, grain size of sand and immersion time were considered. Takahagi sand, Toyoura sand, and Kawasaki clay sieved through a 0.074 mm mesh and seawater were used. The solution, including heavy metal but excluding clay particles, was prepared by the method described above. Samples contained in acrylic cylinders (d = 290 mm, h = 100 mm) were made by mixing small amounts of clay with sand in different ratios ranging from 0.001% to 5%. The solution, including different kinds of heavy metal, was poured into each container. The samples were left in the container for from 30 minutes to 3 hours and the filtrate was analyzed to find the adsorption of heavy metal to clay. Results of these experiments are

(1) Adsorption of mercury increases in proportion to the clay fraction regardless of immersion time and the percentage of adsorption was the same as that in the suspension.

(2) Adsorption of cadmium is 30%-50% regardless of both immersion time and clay fraction because of effects due to seawater.

(3) Adsorption of lead is about 80% regardless of both immersion time and clay fraction.

EFFECTS OF CLAY TYPES ON ADSORPTION OF HEAVY METALS

In this experiment, the difference in adsorption by types of clays was investigated. Six kinds of clay, shown in Table 1, and coral sand from Okinawa were used. For the purpose of a precise comparison, all clays were sieved through 0.074 mm mesh and artificial seawater was used. The experimental method is the same as described for the sand experiments. Results of this experiment are shown in Table 2 and may be described as follows:

(1) Generally, adsorption of heavy metal increases in proportion to turbidity. But the influence of a pH change on adsorption of heavy metal is not clear.

(2) The percentage of mercury adsorbed is higher than that of lead and that of cadmium.

(3) Heavy metal is more easily adsorbed on coral sand than on clay.

TABLE 2. ADSORPTION OF HEAVY METAL TO DIFFERENT KINDS OF CLAY IN JAPAN

			Metal								
			Lead			Cadmium			Mercury		
Clay Type		SS (ppm)	20	200	2000	20	200	2000	20	200	2000
Aomiri	%		-	-	91	-	0	11	36	70	98
	pH		4.3	4.8	5.6	5.2	5.6	5.5	5.4	5.1	5.8
Kawasaki	%		-	6	99	-	6	62	36	93	90
	pH		4.1	4.9	6.6	5.5	6.2	6.9	5.5	6.3	6.8
Nagoya	%		0	-	84	-	1	32	2	90	97
	pH		4.2	4.4	6.0	5.1	5.4	5.3	5.4	5.8	6.5
Fushiki-Toyama	%		-	-	46	1	4	5	10	95	85
	pH		4.3	4.6	5.8	5.1	5.5	6.2	5.1	5.7	6.3
Kobe	%		-	3	99	0	11	83	9	93	98
	pH		4.0	5.0	7.2	5.3	6.4	6.3	5.3	6.4	6.4
Moji	%		-	-	97	2	0	40	32	92	92
	pH		4.0	4.4	6.7	5.3	6.1	7.0	5.3	6.1	7.1
Okinawa (coral sand)	%		-	88	99	7	12	74	94	89	70
	pH		6.5	9.3	9.5	9.0	9.6	9.7	8.1	9.6	9.7

EXPERIMENTS ON SAND FILTRATION

FILTERING EFFECT ON BACKFILL SAND

The absorption experiments show that heavy metal is adsorbed to clay and that the percentage of adsorption depends on the kind of clay, kind of water, pH of the solution, turbidity and other factors. If the clay particles, including heavy metal and other contaminated substances, remain in the backfill sand while water permeates through the sand, secondary pollution in the vicinity can be minimized.

The purpose of this experiment was to investigate the filtering capacity of sands used for backfilling. The experimental apparatus consists of two cylinders as shown in Figure 9. Kawasaki clay, Takahagi sand, and Toyoura sand were used for the filter media. Factors such as concentration of the suspension, water pressure, thickness of sand, kind of sand and kind of water were tested. The results of this experiment are

(1) The filtering effect of the sand changed with time until the sand finally became impermeable due to clogging by clay (Figure 10).

(2) In the Toyoura sand, which has a small grain size, the filtering effect is nearly total.

(3) The clay particles flocculate in seawater and thus the filtering effect of sand becomes more pronounced than in freshwater.

(4) The most important factor seems to be the grain size of sands, that is, if the grain size is smaller, the filtering effect is more significant.

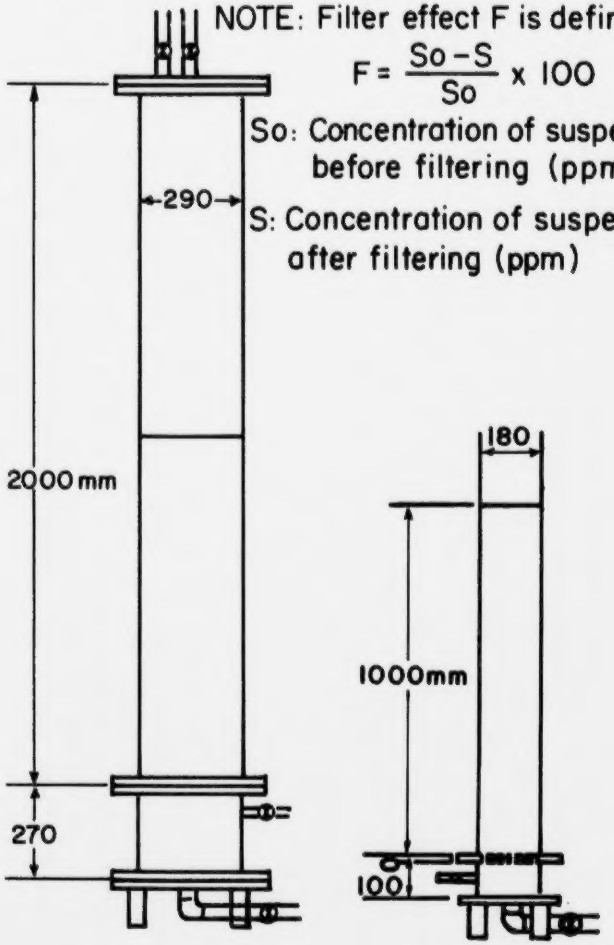
(5) Any noticeable difference in filtering effects due to factors such as water pressure, thickness of sand layer and concentration of the suspension was not found.

EFFECTS OF GRAIN SIZE OF SANDS ON FILTRATION

The purpose of this experiment was to investigate the effect of sand grain size on the efficiency of filtering solutions with heavy metals. In preparing test samples, careful attention was paid to adjusting the grain size of sands and clays. Kawasaki clay, Takahagi sand and Toyoura sand were carefully sieved. A cylinder (d = 10 cm, h = 70 cm) was kept at 500 ppm and the water head was kept at 60 cm. The filtration time was 1 hour. In considering the relative influence of grain size of sands to that of clays, the clay-sand ratio, R (d/D_{10}) is defined as the ratio of the maximum grain size of clay, d = 0.074 mm, to the effective grain size of sand, D_{10}. The experiment was carried out by changing this ratio, R. Figure 11 shows the results of these experiments and from this it is apparent that the filtering effect does depend on the clay-sand ratio, R, and is about 100% effective when $R \geq 0.2$.

PERMEABILITY TESTING OF SEALS FOR PILING JOINTS

In Japan, several kinds of seals have been used to make the joints between sheet piles watertight. These seals are classified into four types: mortar, asphalt, urethane and cellulose. Since information on the permeability of conventional mortar seals is available, the permeability tests were performed on the other three types of seals. Urethane and cellulose work as sealing materials by swelling. Asphalt simply fills the joints.



NOTE: Filter effect F is defined as

$$F = \frac{So - S}{So} \times 100$$

So: Concentration of suspension before filtering (ppm)

S: Concentration of suspension after filtering (ppm)

Figure 9. Cylinder used for sand filtration experiment.

TEST NO.	SAND	WATER	SS (ppm)	HEAD (cm)
1	TAKAHAGI	FRESH WATER	1500	250
2	"	SEA WATER	2500	250
3	"	FRESH WATER	3000	400
4	"	"	500	250
5	TOYOURA	"	1000	250

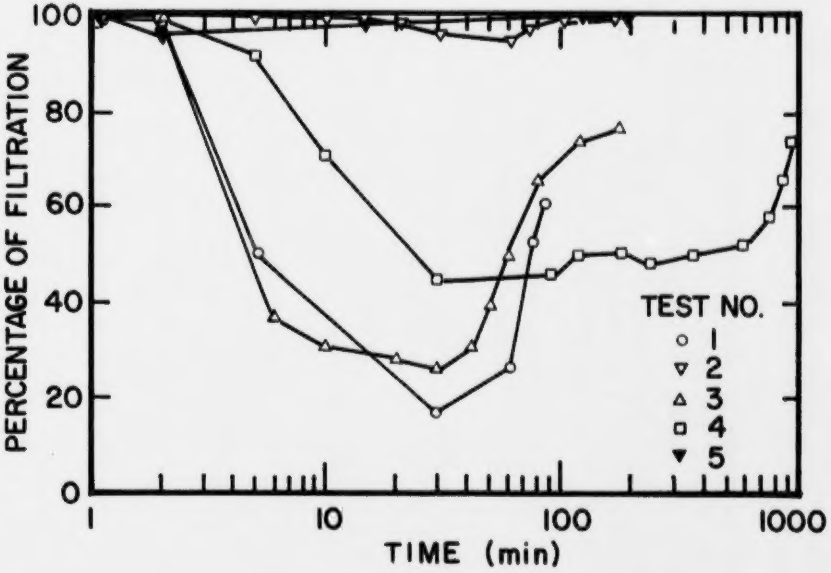


Figure 10. Filtering effect of sand.

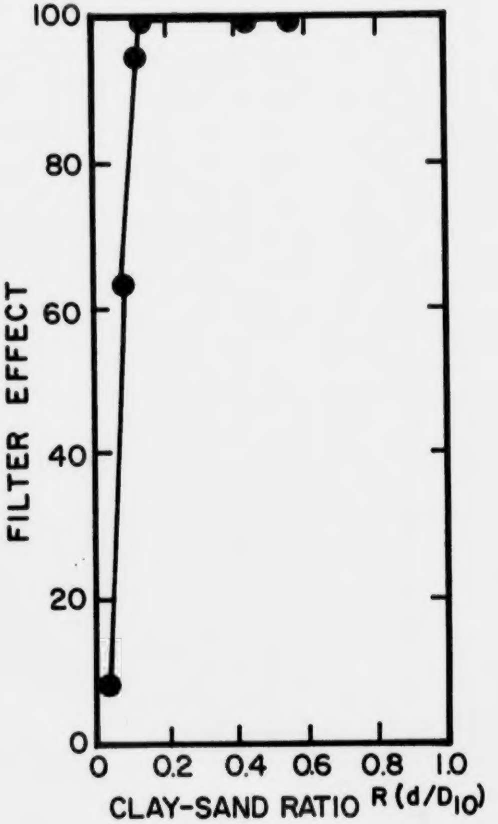


Figure 11. Relation between filtering effect and clay-sand ratio.

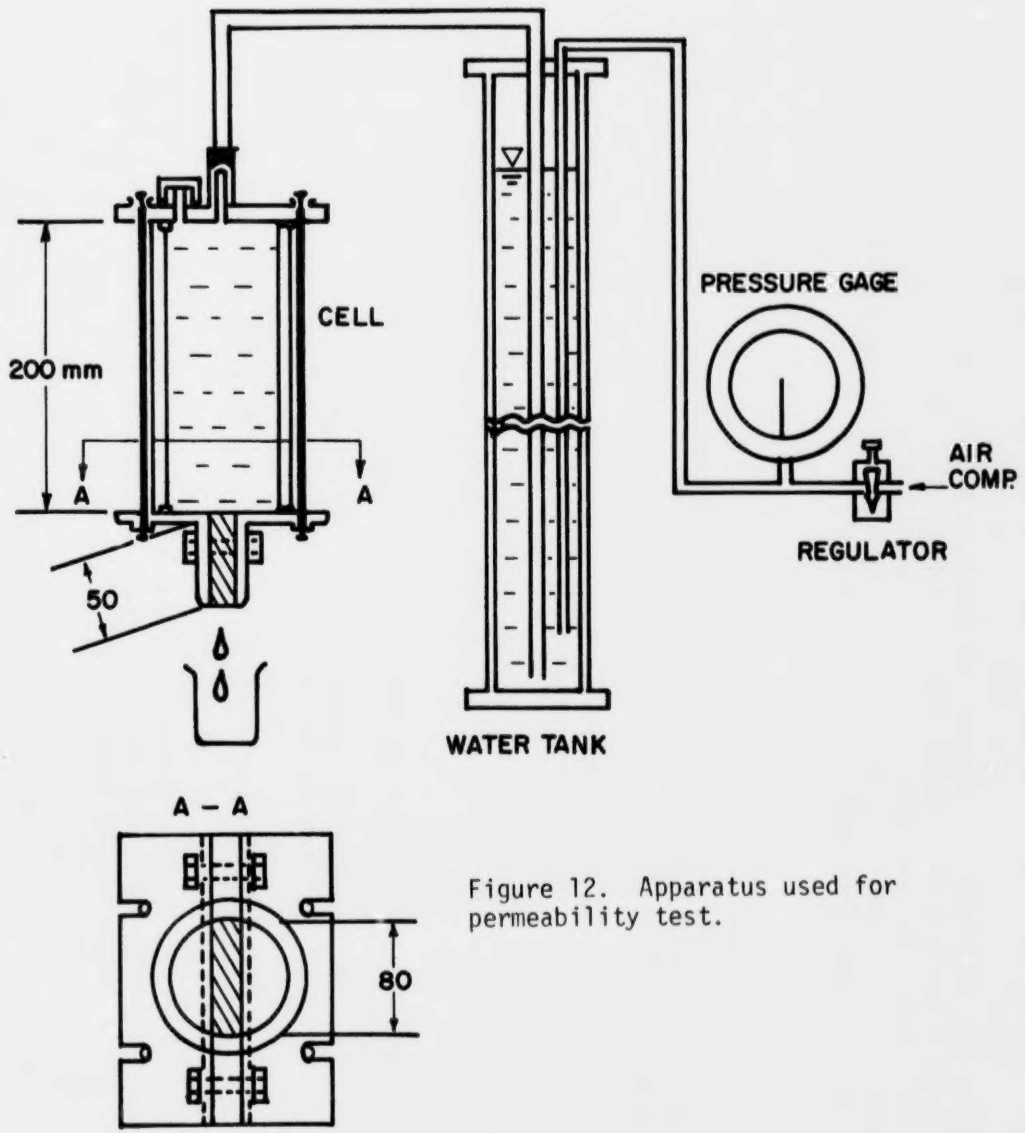

CELL
200 mm
A
A
50
WATER TANK
PRESSURE GAGE
AIR COMP.
REGULATOR
A – A
80

Figure 12. Apparatus used for permeability test.

The apparatus and results are shown in Figures 12 and 13. From Figure 13 it is clear that the coefficient of permeability, K, is on the order of 10^{-3} N 10^{-4} cm/sec and increases in proportion to the water pressure. For practical use of these results, further investigations on the durability of seals with hot water and other substances are necessary.

CONCLUSIONS

Several series of experiments were conducted to investigate the filtering effect of containment walls for landfills with dredged materials. From the experimental results, it was concluded that:

1) Adsorption of heavy metals (such as mercury, cadmium, and lead) to clay increases in proportion to turbidity.

2) Adsorption of heavy metal to clay is greater in alkaline solutions than in acid solutions and this tendency is more notable for cadmium and lead.

3) Cadmium is not adsorbed on clay in seawater.

4) Clay particles produce flocs in seawater and are deposited more rapidly than in freshwater.

5) Heavy metal is adsorbed on clay in sands in the same way as in suspension.

6) Clay samples taken in the main ports of Japan had similar physical and chemical properties; hence they showed a similar tendency toward adsorption of heavy metals.

7) The filtering effect of sand depends on the clay-sand ratio, R, defined as the ratio of the maximum grain size of clays, $d = 0.074$ mm to the effective grain size of sands, D_{10} and is nearly 100% for $R \geq 0.2$.

8) It is expected that backfill sand has a filtering effect which prevents the permeation of contaminated substances.

9) Seals used for making watertight joints between sheet piles are useful in preventing permeation. When seals are used the coefficient of permeability, K, is on the order of 10^{-3} N 10^{-4} cm/sec.

Although useful data were obtained on prevention of permeation of contaminated substances, it is necessary to continue the research because many problems remain unsolved regarding long-term permeation, the permeation velocity of contaminated substances and the influence of tides and earthquakes.

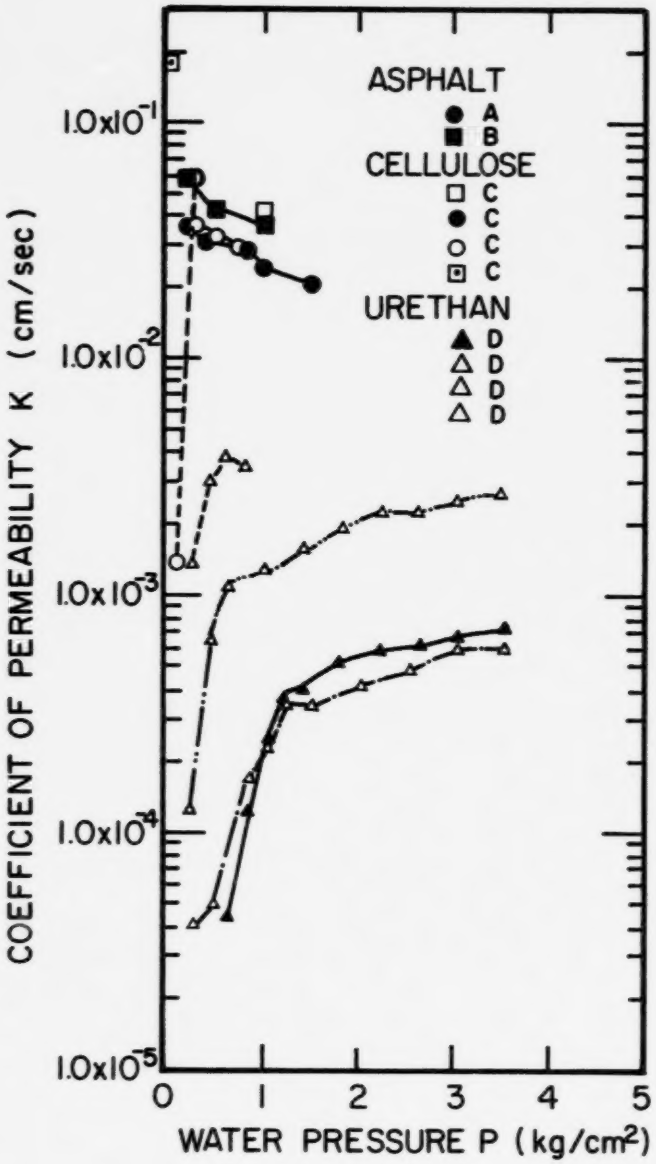


Figure 13. Relation between coefficient of permeability K and water pressure.

ACCUMULATION OF MERCURY BY FISH FROM CONTAMINATED SEDIMENTS

R. Hirota,[1] M. Fujiki,[2] Y. Ikegaki[3] and S. Tajima[2]

ABSTRACT

The relationship between body length and mercury concentration in fish is not known for fish that inhabit areas of the sea bottom sediments containing mercury. Fish captured from Minimata Bay where there is a great deal of mercury pollution and uncontaminated fish grown in pens in Minamata Bay were studied to determine if there was a correlation between accumulation of mercury in fish flesh and rearing time in Minamata Bay.

The correlation coefficients (r) between the concentration of accumulated mercury and body length were 0.676 for common seabass, 0.65 for marbled rockfish, 0.63 for jack mackerel, 0.61 for black sea bream, and 0.35 for rockfish. These are considered statistically significant. Correlation coefficients of 0.3 for red sea bream, 0.096 for the common nibbler, and -0.06 for mullet were not statistically significant.

Fish were separated into four groups and reared in pens at two stations in Minamata Bay. The tendency for mercury accumulation in the fish from the different experimental groups was different for each species.

From the results it appears that selected species and ages of fish should be used when a mercury monitoring program is implemented in Minamata Bay.

INTRODUCTION

More than seven years have passed since a chemical factory, using mercury as a catalyst, stopped discharging its effluent into Minamata Bay. However, the bottom sediment of the bay still contains mercury. It is estimated that over 25 ppm of mercury—more than 1.5 million m^3 contaminates the bay's sediments. The highest concentration of mercury has been reported at 262 ppm.

It is known that the mercury accumulated in fish gradually increases in proportion to body weight as the fish grows older. However, no investigation of the exact relationship between the concentration in the fish and the body

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[1] Aitsu Marine Biological Station, Kumamoto University, Kumamoto, Japan.
[2] Institute of Community Medicine, the University of Tsukuba, Ibaraki, Japan.
[3] Assistant Chief, Environmental Department, Kumamoto Prefectural Government, Japan.

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weight had been done for fish living in the sea where the bottom sediment contained mercury, such as in Minamata Bay.

Since 1973 the government of Kumamoto Prefecture has investigated the concentration of mercury in fish captured from Minamata Bay, and it has also, since 1975, been investigating the relationship between the mercury concentration in red sea bream, common nibbler and marbled rockfish and rearing time for fish reared in the pens at Minamata Bay.

This paper reports on the results obtained from statistical analysis of the data derived from these investigations.

MATERIALS AND METHODS

INVESTIGATION A

Fish captured from Minamata Bay were used for the investigation. The body length and concentration of mercury in the muscle of fish were measured. Flameless atomic absorption spectrophotometric techniques were used to analyze the total mercury in the fish muscle.

Rockfish, red sea bream, common nibbler, mullet, common seabass, marbled rockfish, jack mackerel and black sea bream were selected as test species from the many kinds of fish—mostly because these were captured in sufficient numbers for statistical analysis. Correlation coefficients (r-values) between the concentration of the accumulated mercury and body length were determined for the above eight species of fish.

INVESTIGATION B

Red sea bream grown from eggs in a fish farm at Oyano Marine Station in Kumamoto Prefecture and marbled rockfish and common nibbler captured in the field were used in the investigation. The fish were separated into four groups and were reared in pens at the two stations in Minamata Bay shown in Figure 1. The rearing period was as follows: Group A: red sea bream and marbled rockfish were reared from July 1975 to December 1975. Group B: common nibbler were reared from December 1976 to May 1977. Group C: red sea bream and common nibbler were reared from June 1977 to November 1977. Group D: red sea bream and common nibbler were reared from November 1977 to May 1978.

Ten fish from each group were taken from the pen every 10 days during the rearing period and the concentration of total mercury in the muscle tissue was measured using a flameless atomic absorption spectrophotometer. Using these data, the relationship between rearing time and accumulated mercury was studied.

RESULTS

The results of Investigation A are in Table 1. The concentration of accumulated mercury in common seabass and marbled rockfish showed 0.41 ppm

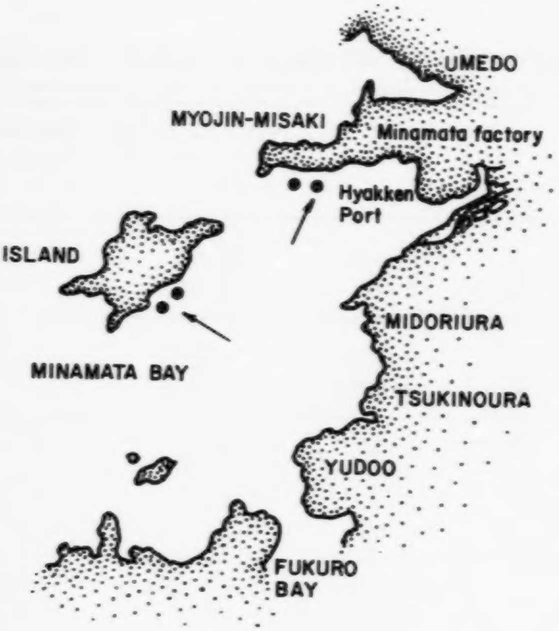

Figure 1. Rearing pen sites in Minamata Bay.

(wet weight) and 0.85 ppm (wet weight), respectively. These values were higher than the Japanese government's provisional criteria for mercury in fish of 0.4 ppm (wet weight).

The statistically significant correlation coefficients between accumulated mercury and body length were 0.676 for common seabass ($p << 0.001$), 0.65 for marbled rockfish ($p << 0.001$), 0.63 for jack mackerel ($p << 0.001$, 0.61 for black sea bream ($p << 0.001$) and 0.35 for rockfish ($p < 0.001$). Correlation coefficients (r-values) of 0.3 for red sea bream ($p < 0.1$), 0.096 for common nibbler ($p < 0.3$) and -0.06 for mullet ($p < 0.6$) were not statistically significant.

For the eight species of the fish, scatter diagrams for the concentration of accumulated mercury vs body length are given in Figures 2 to 9.*

The results of Investigation B are shown in Figures 10 to 13. For group A, the concentration of mercury in red sea bream increased gradually, although that of marbled rockfish did not increase during the rearing period. For group B, the concentration of mercury in the common nibbler increased a little. For group C, the concentration of mercury in red sea bream and common nibbler did not increase. And for group D, the concentration of mercury in red sea bream and common nibbler increased gradually.

DISCUSSION

It is important to monitor water quality so no further pollution of the marine environment is caused by the dredging which will clean up pollution in Minamata Bay. If mercury elutes from the bottom sediment to the marine water,

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*Figures 2 through 13 are found at the end of the text, beginning on page 230.

TABLE 1. CORRELATION OF ACCUMULATED MERCURY AND BODY LENGTH IN FISH

		Rockfish	Red sea bream	Common nibbler	Mullet
	n	121	27	129	85
Body length (cm)	$\bar{x}$	15.615	12.9	18.767	32.66
	Sx	2.51	4.51	3.760	5.200
Total mercury (ppm)	$\bar{y}$	0.2255	0.231	0.1256	0.045
	Sy	0.0850	0.135	0.043	0.032
	r	0.35	0.3	0.097	-0.06
	t	4.093	1.778	1.100	0.529
	d.f.	119	25	127	83
	P	<0.001	<0.1	<0.3	<0.6
Regression line		$y=0.012x + 0.04$	$y=0.0009x + 0.23$	$y=0.0011x + 0.105$	$y=-0.00004x + 0.045$

		Common seabass	Marbled rockfish	Jack mackerel	Black sea bream
	n	114	133	133	97
Body length (cm)	$\bar{x}$	36.803	14.747	21.604	21.89
	Sx	11.034	2.29	6.309	5.228
Total mercury (cm)	$\bar{y}$	0.4125	0.8488	0.1379	0.384
	Sy	0.3561	0.3179	0.0938	0.4308
	r	0.676	0.65	0.63	0.610
	t	9.712	9.842	9.236	7.497
	d.f.	112	131	131	95
	P	<<0.001	<<0.001	<<0.001	<<0.001
Regression line		$y=0.0217x - 0.387$	$y=0.091x - 0.5$	$y=0.0093x - 0.06$	$y=0.0495x - 0.70$

or if the concentration of mercury in marine food products is increasing, the dredging must be slowed or stopped until the cause of the pollution is determined. To support the monitoring plan, the baseline condition of fish in Minamata Bay must be established before dredging begins.

The results presented in this paper show that the concentration of mercury in common seabass, marbled rockfish and some of the black sea bream was higher than the Japanese government's provisional criteria for mercury in fish. This suggests that Minamata Bay is still polluted by mercury when compared to other areas.

Correlations between the concentration of accumulated mercury and body length were statistically significant for common seabass, marbled rockfish, jack mackerel, black sea bream and rockfish. The concentration of mercury in the above five species of fish increases in proportion to the age of the fish. The presumed ages of common seabass, marbled rockfish and jack mackerel were 1 to 7 years old and black sea bream was 1 to 5 years old for those fish used in the investigation. The correlation coefficients between body length and concentration of accumulated mercury in red sea bream, common nibbler and mullet were not statistically significant. This may be because the presumed age of red sea bream was 1 year old. Another reason for the lack of correlation between the concentration of accumulated mercury and body length is that rates of mercury uptake in common nibbler and mullet are low and rates of excretion from these fish are high.

The studies of accumulation vs rearing time showed no relationship between the experimental groupings of fish. The accumulation of mercury in fish may be lower than the statistical deviation of the level of concentration of mercury among fish because the rearing period was only 6 months. There was a slight accumulation exhibited by the marbled rockfish and common nibbler.

CONCLUSION

This investigation suggested that certain species and ages of fish should be identified as bio-monitors to signal any increased pollution in Minamata Bay. Jack mackerel and black sea bream would be good species for a monitoring effort because these have the highest correlation between body length and the concentration of accumulated mercury in muscle tissue. Also, the present concentration of mercury in muscle tissue of these fish is below the government criterion of 0.4 ppm.

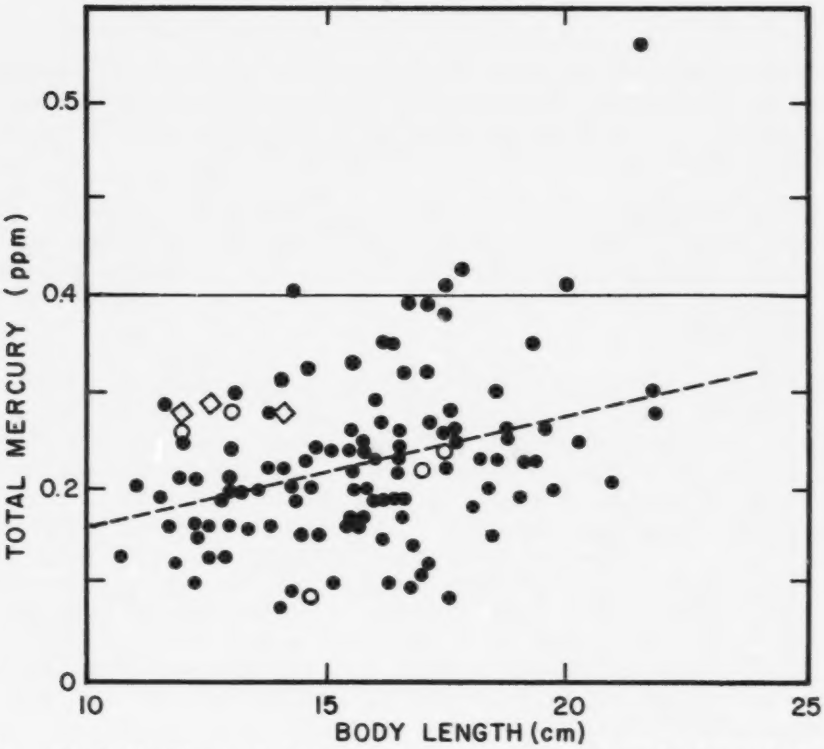

Figure 2 . Distribution of accumulated mercury and body length for rockfish.

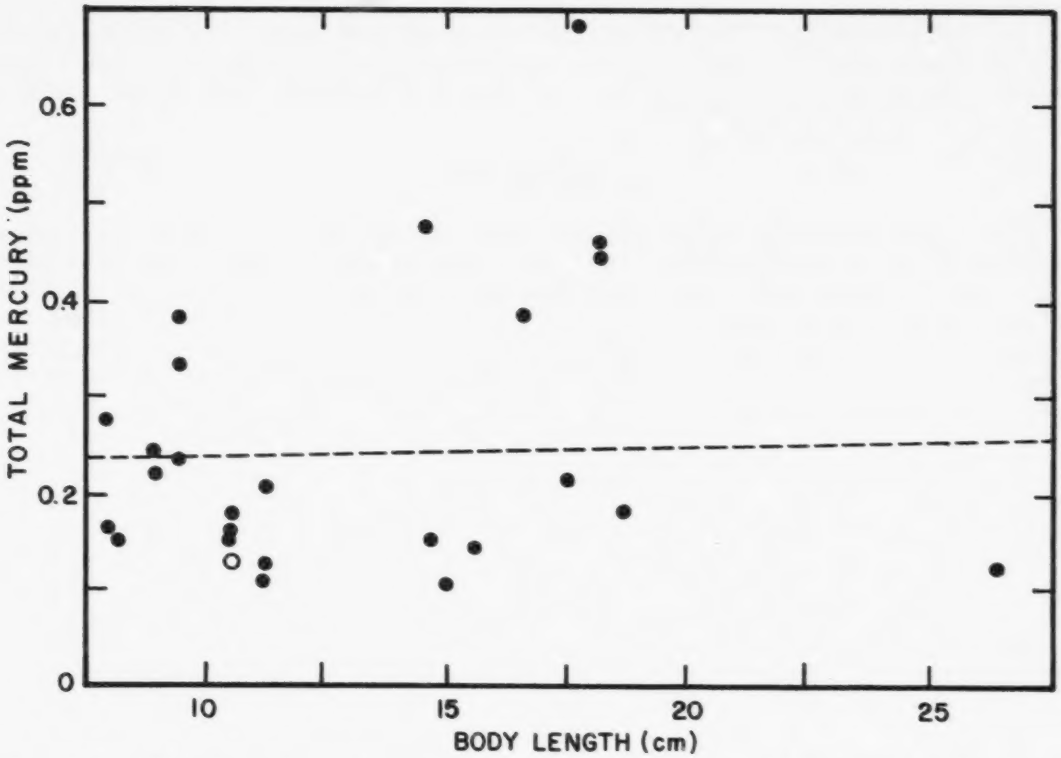

Figure 3. Distribution of accumulated mercury and body length for red sea bream .

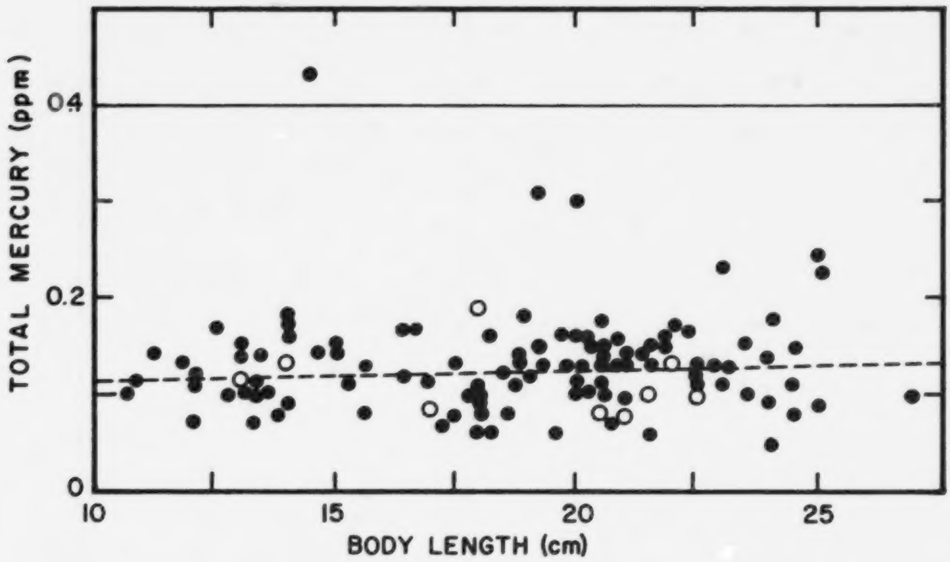

Figure 4. Distribution of accumulated mercury and body length for common nibbler.

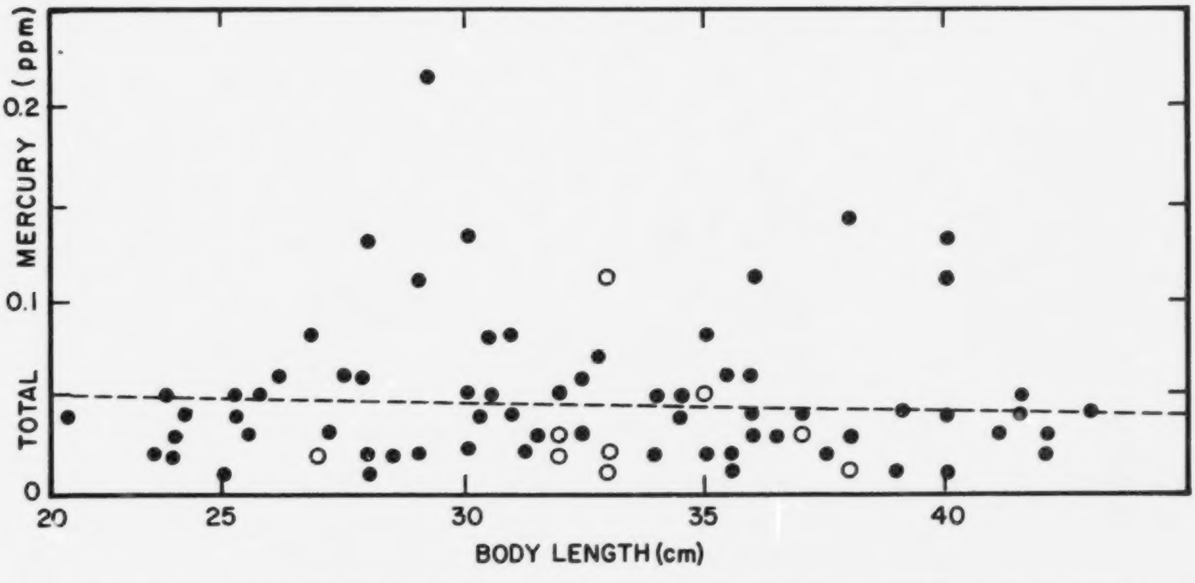

Figure 5. Distribution of accumulated mercury and body length for mullet.

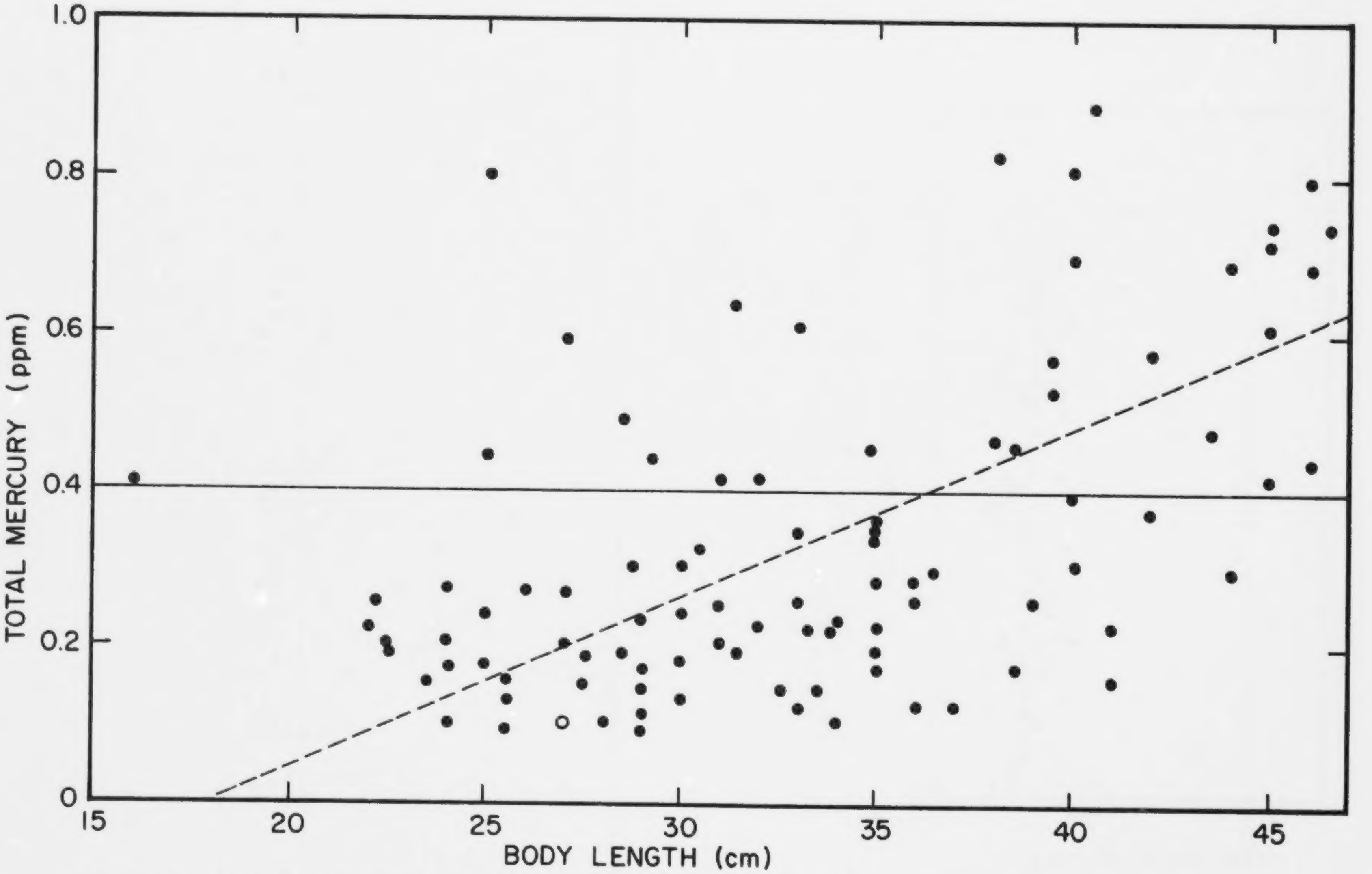

Figure 6a. Distribution of accumulated mercury and body length for common sea bass.

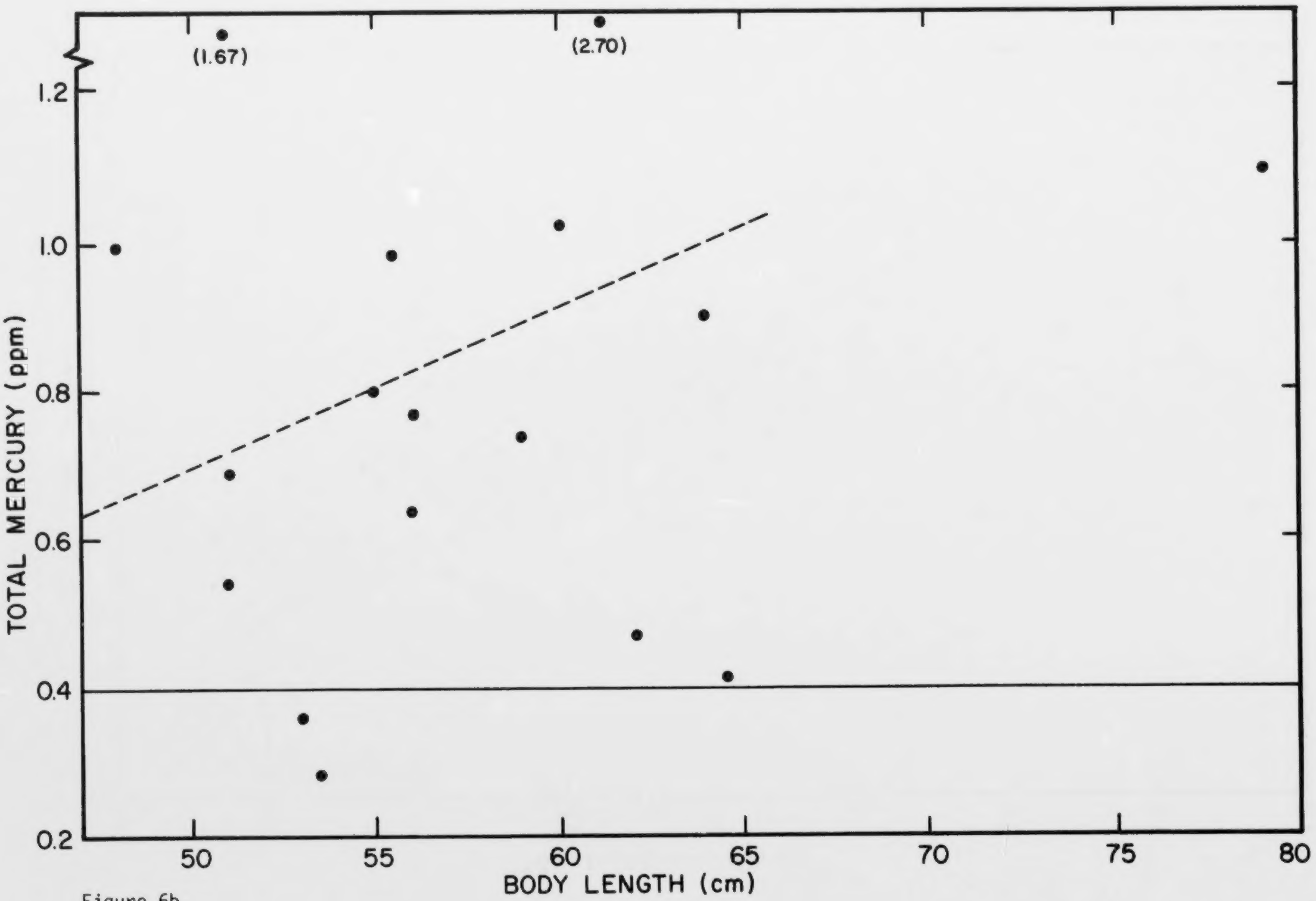
(1.67)
(2.70)
1.2
1.0
0.8
0.6
0.4
0.2
TOTAL MERCURY (ppm)
50
55
60
65
70
75
80
BODY LENGTH (cm)

Figure 6b.

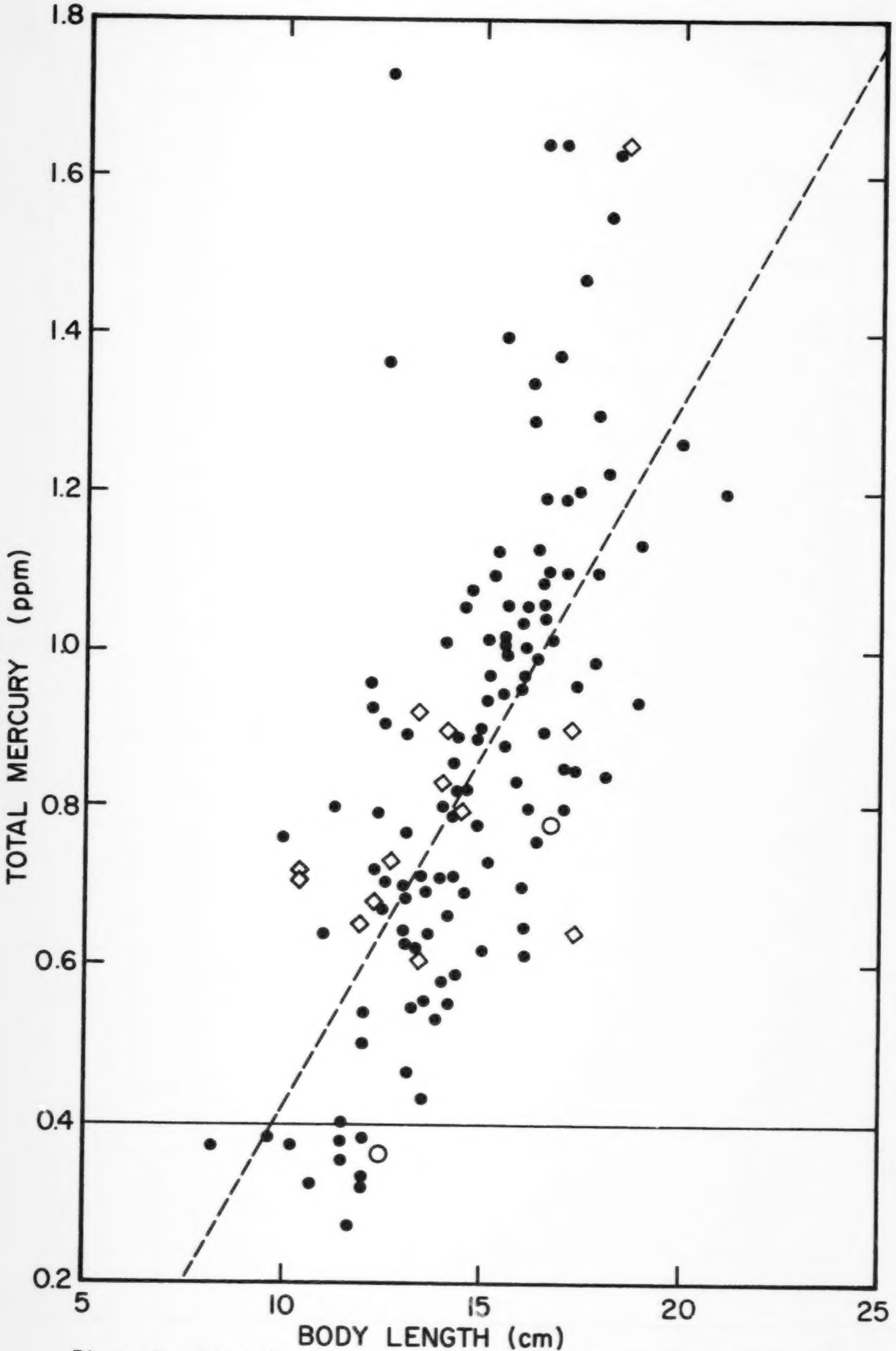

Figure 7. Distribution of accumulated mercury and body length for marbled rockfish.

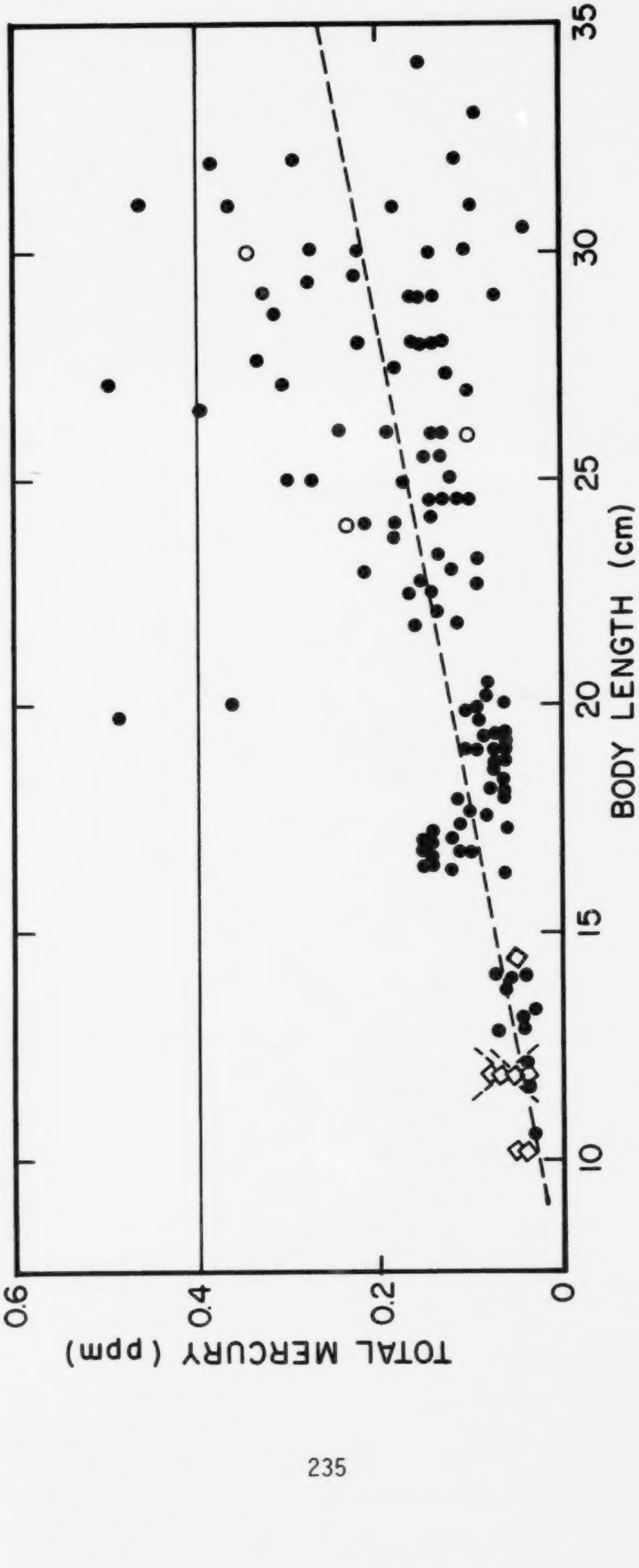

Figure 8. Distribution of accumulated mercury and body length for jack mackerel.

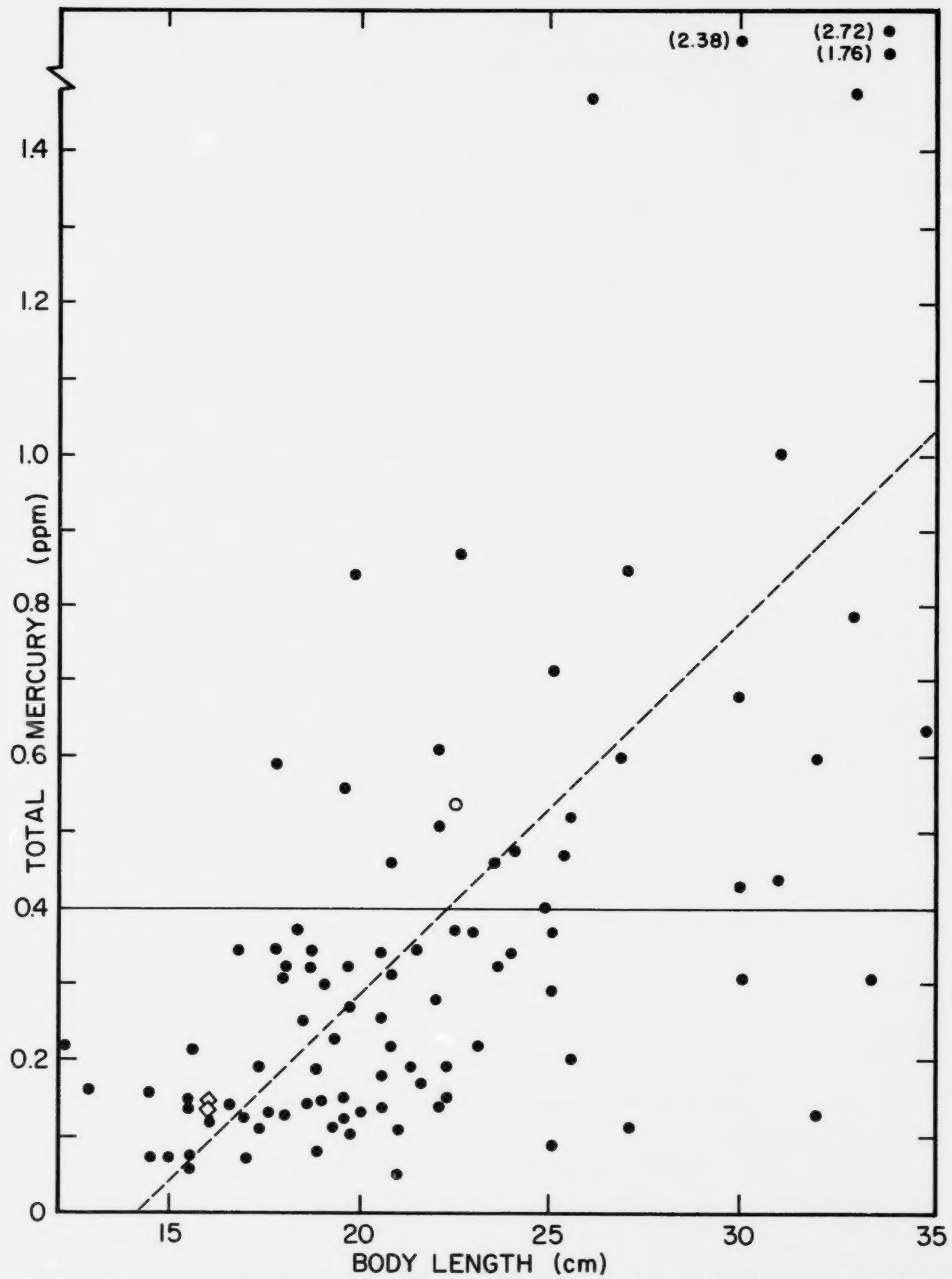

Figure 9. Distribution of accumulated mercury and body length for black sea bream.

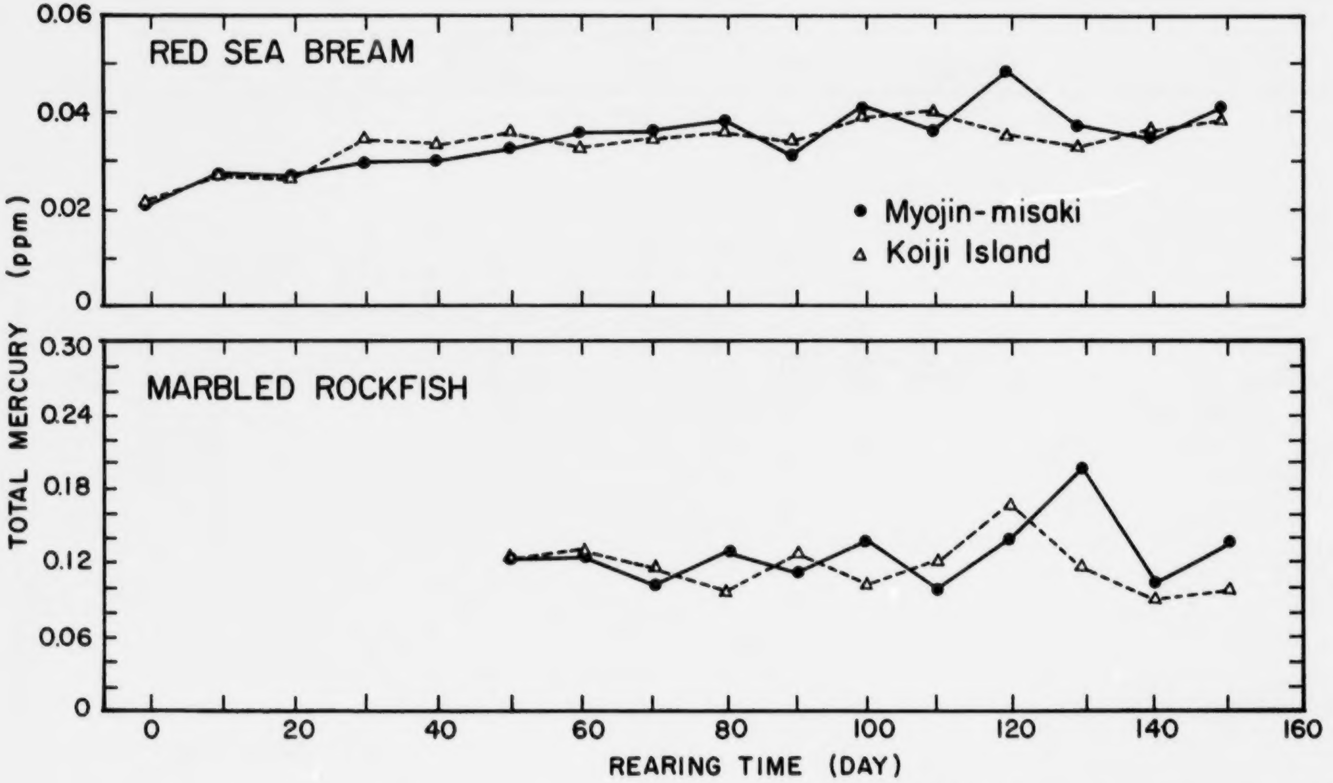

Figure 10. Relationship between accumulated mercury in fish and rearing time--group A.

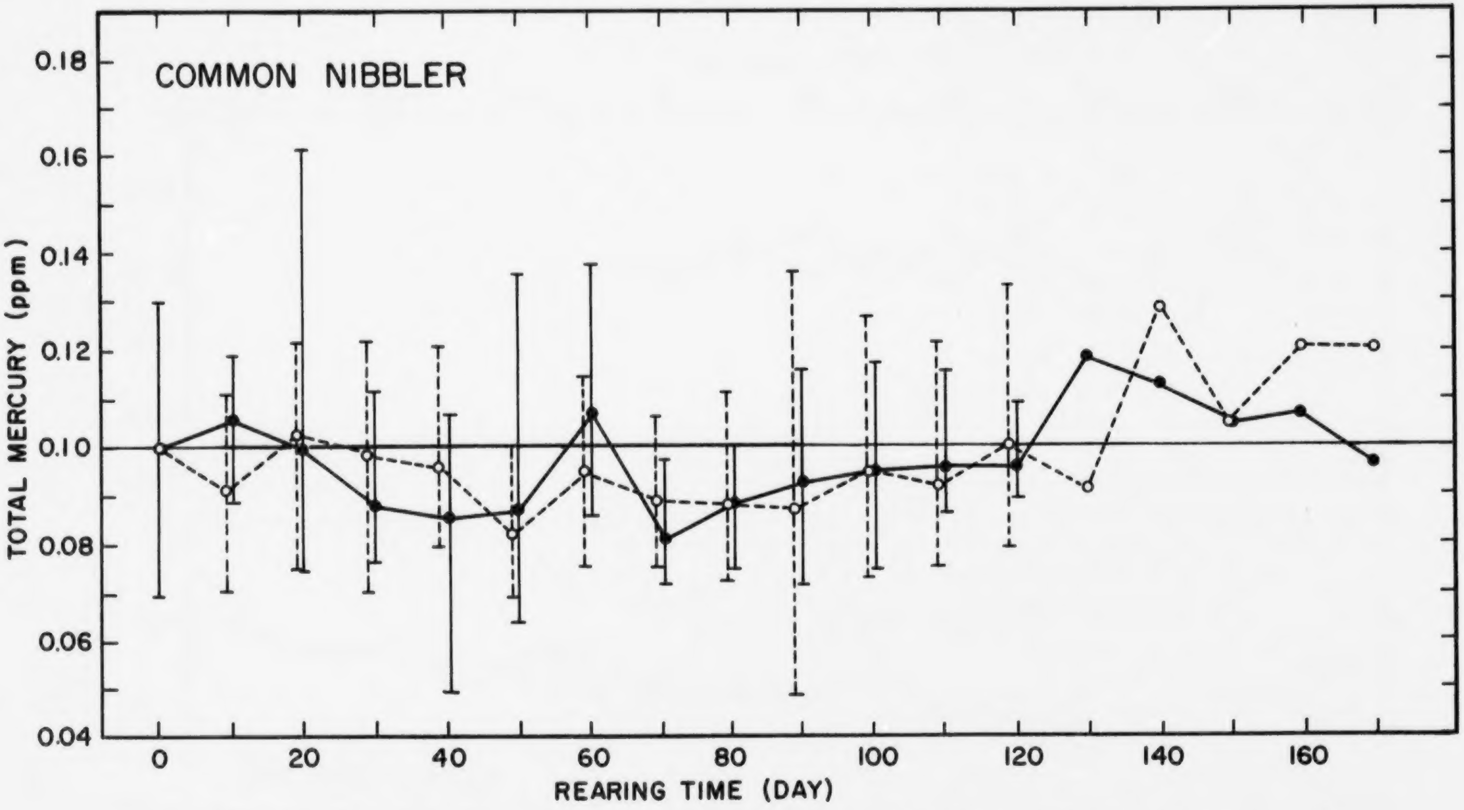

Figure 11. Relationship between accumulated mercury in fish and rearing time--group B.

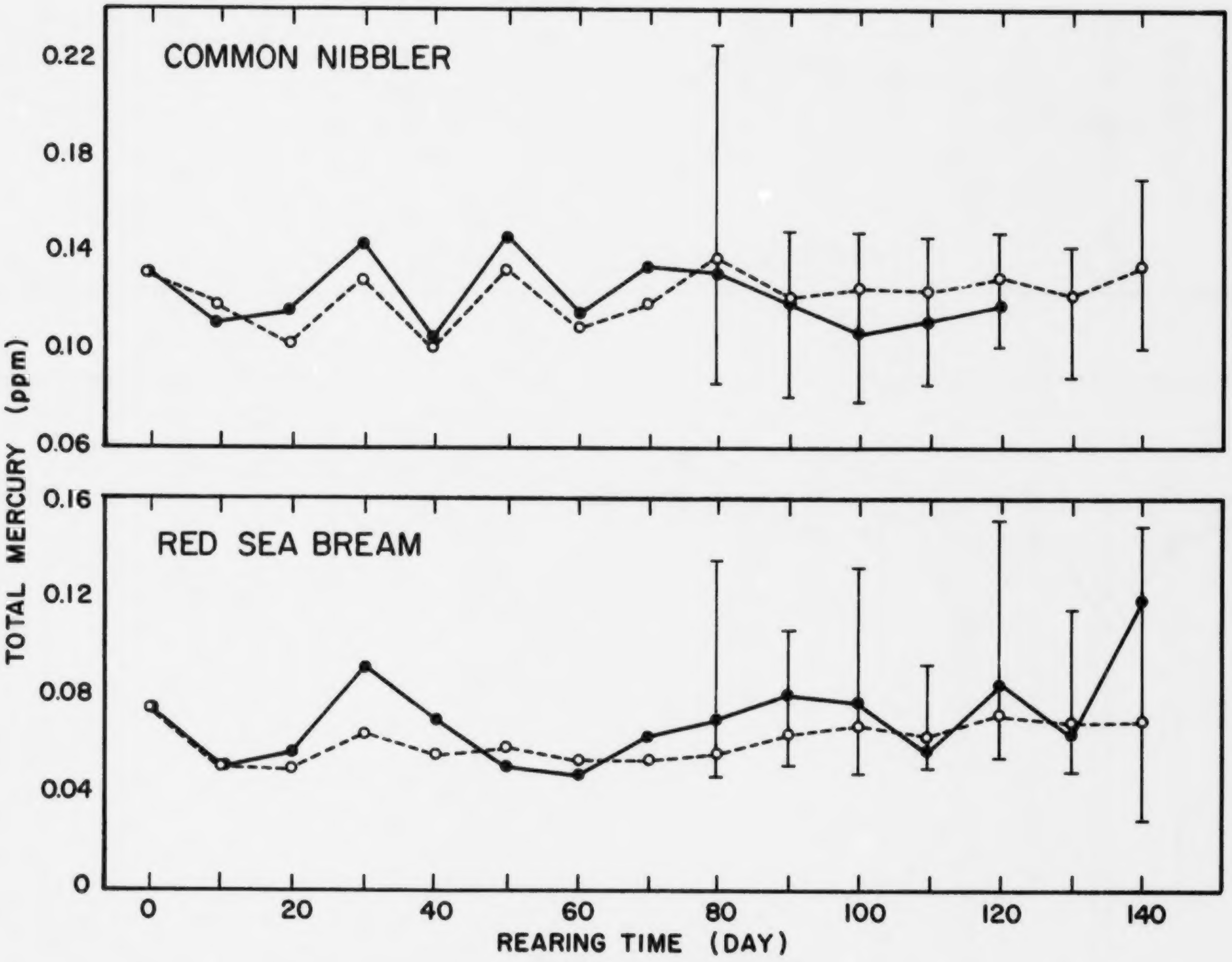

Figure 12. Relationship between accumulated mercury in fish and rearing time--group C.

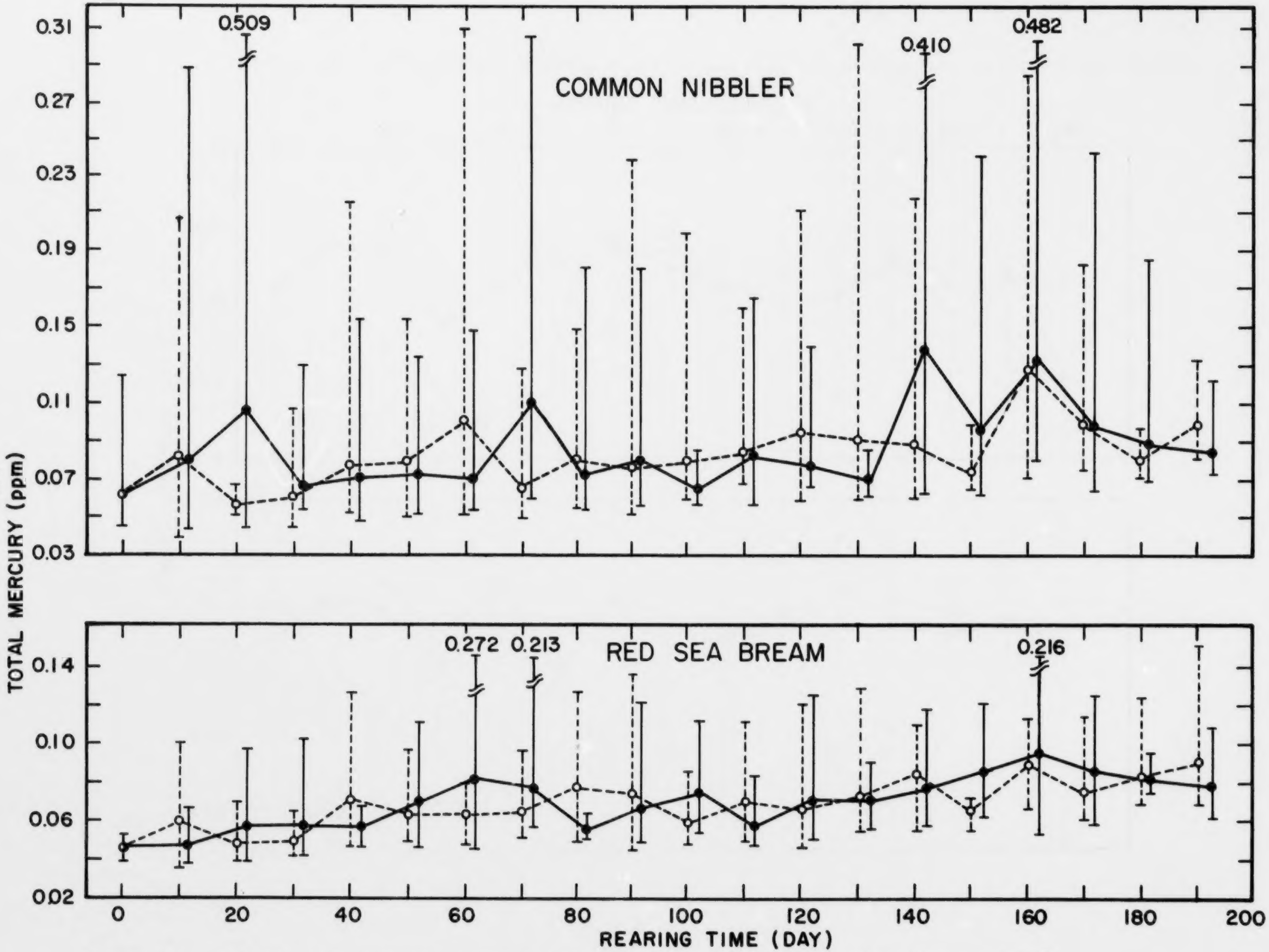

Figure 13. Relationship between accumulated mercury in fish and rearing time--group D.

APPROACHES FOR MITIGATING THE KEPONE CONTAMINATION IN THE HOPEWELL/JAMES RIVER AREA OF VIRGINIA

K. M. Mackenthun, M. W. Brossman,
J. A. Kohler, C.R. Terrell
Criteria and Standards Division
Office of Water and Waste Management
U.S. Environmental Protection Agency
Washington, D.C. 20460

ABSTRACT

The Kepone Mitigation Feasibility Project discussed at the third U.S./Japan meeting on management of bottom sediments containing toxic substances has been completed. The four-volume project report was forwarded to the Governors of Virginia and Maryland, together with a recommendation of formation of a task force to consider and implement the report's recommendations.

This paper summarizes the nature of the contamination found in the Hopewell/James River area and describes the mitigation approaches evaluated and the mitigation actions recommended.

Extensive sampling efforts revealed the highly persistent Kepone contamination remains on the land in Hopewell, in sewage lines and streams and in the James River. Accordingly, mitigation methods had to focus on the problems of land, water and sediment contamination. Conventional (dredging), non-conventional, and degradation/biological approaches to mitigation were investigated.

The investigation of conventional mitigation/removal approaches included an analysis of world-wide dredging techniques, with an evaluation of the most promising dredging techniques for removal of contaminated sediments from specific sites, engineering studies to contain, stabilize or remove contaminated sediments at points of inflow into the James River and evaluation of the engineering requirements for removal of Kepone-contaminated sediments from the James River, assessment of dredge spoil sites, fixation of the dredge spoil and treatment of the elutriate.

A wide range of non-conventional mitigation approaches was evaluated for dredge spoil fixation, elutriate treatment, in situ stabilization, and isolation. Approaches ranged from silicate, organic- and sulfur-based fixation agents through use of retrievable and non-retrievable sorbents which would take up Kepone from the contaminated sediments and water.

Natural degradation and biological mitigation approaches were examined concurrently with engineering approaches. However, none of these approaches shows high promise for Kepone mitigation at this time.

As a result of these investigations, a number of immediate mitigation actions were recommended in addition to further development of promising mitigation techniques. Among the recommendations was the adaptation of Japanese dredging and sediment fixation technology. In addition, the investigative techniques and findings were recommended for solution of other similar waterway contamination problems in the United States.

INTRODUCTION

At the third U.S./Japan meeting on management of bottom sediments containing toxic substances a paper was presented, "Mitigation Feasibility for the Kepone-Contaminated James River, Virginia" (1). That paper described the origin of the Hopewell/Virginia Kepone contamination, the study approach to assess the problem, preliminary findings on the nature and extent of the contamination and the specific conventional (dredging) and unconventional approaches under investigation to mitigate the problem. The study is now completed and the four-volume project report has been provided to the Governors of Viginia and Maryland.

This paper is devoted primarily to characterizing the nature of the contamination found, techniques evaluated for mitigation of the contamination problem and the specific recommended mitigation actions. Additional details may be found in the Kepone Mitigation Project Report (2).

NATURE OF THE CONTAMINATION

Kepone, a highly chlorinated hydrocarbon pesticide, was discharged into the environment around Hopewell, Virginia from 1966 to 1975 from two manufacturing operations. The Allied Chemical Corporation's Semi-Works Plant produced Kepone intermittently from 1966 to 1974. Life Science Products Company began Kepone production under contract to Allied Chemical in 1974 and continued production until closure of the plant in September 1975. Fish and sediment samples indicate Kepone contamination existed in the James River as early as 1967. Figure 1 shows the principal landmarks around the Hopewell area which will be cited in the discussion of the contamination and remedial actions.

Early warnings of Life Science Products careless manufacturing and disposal practices were apparent with the malfunctioning of the digesters of the Hopewell sewage treatment plant and the deleterious health effects on the production workers. Subsequently, the finding of high levels of Kepone contamination in James River fish brought about a ban on fishing for a wide range of species. The releases from the Life Science Products plant into the environment were associated with atmospheric emissions, wastewater discharges and bulk disposal of off-specification batches. The atmospheric emissions from the plant settled on the surface soils. Wastewater discharges entering the sewage system passed through the Hopewell sewage treatment plant into Bailey Creek, passing into Bailey Bay and the James River. Disposal of off-specification batches and manufacturing residues of Kepone occurred at a minimum of two sites-the Hopewell landfill and the disposal trench at the former Pebbled Ammonium Nitrate Plant. Following closure of Life Science Products, residues from the dismantled plant and site cleanup were buried at the landfill.

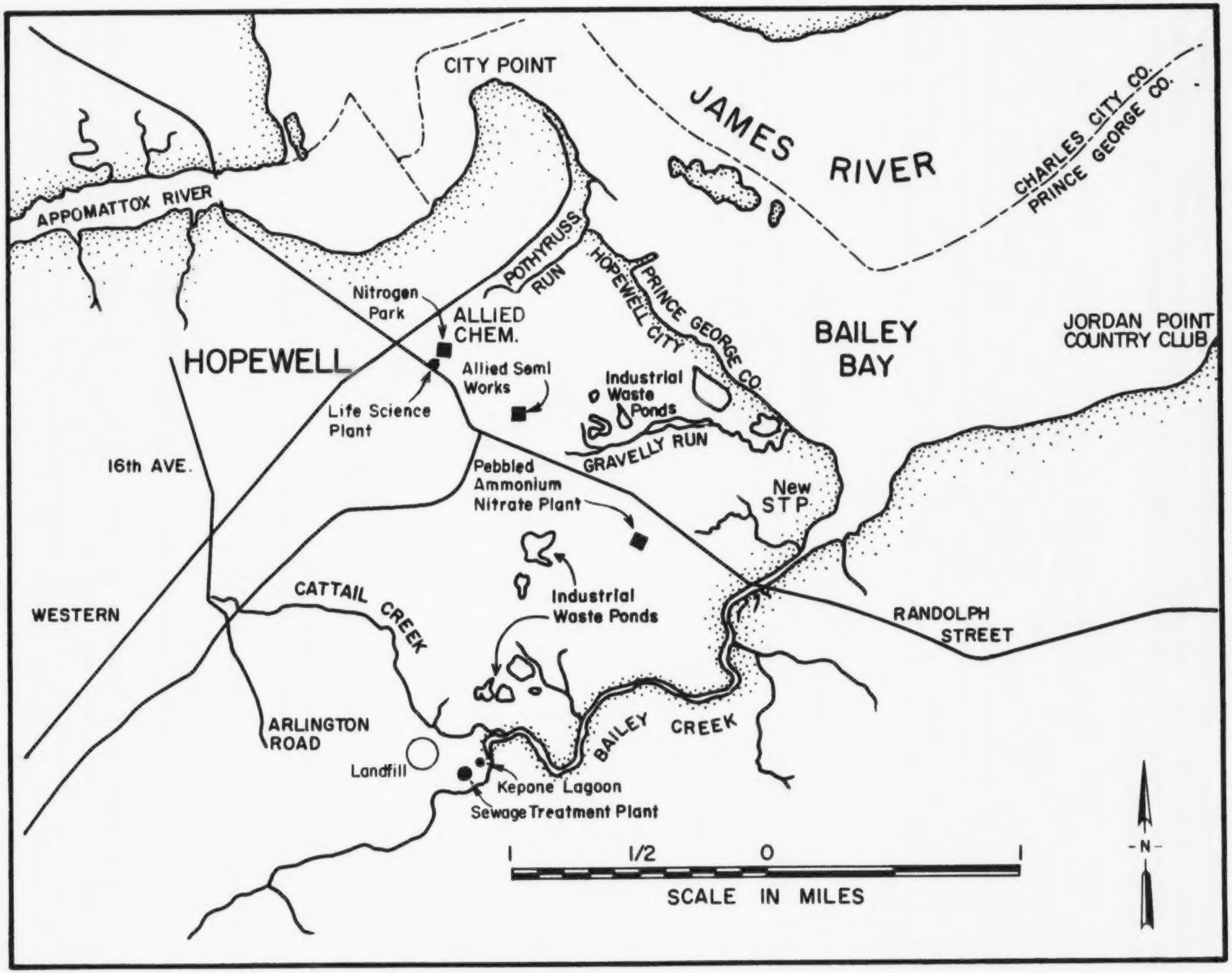
CITY POINT
JAMES RIVER
CHARLES CITY CO.
PRINCE GEORGE CO.
APPOMATTOX RIVER
POTHYRUSS RUN
HOPEWELL CITY
PRINCE GEORGE CO.
BAILEY BAY
JORDAN POINT COUNTRY CLUB
Nitrogen Park
ALLIED CHEM.
HOPEWELL
Allied Semi Works
Industrial Waste Ponds
Life Science Plant
GRAVELLY RUN
16th AVE.
Pebbled Ammonium Nitrate Plant
New STP
CATTAIL CREEK
Industrial Waste Ponds
WESTERN
RANDOLPH STREET
ARLINGTON ROAD
BAILEY CREEK
Landfill
Kepone Lagoon
Sewage Treatment Plant
1
1/2
0
1
SCALE IN MILES
N

Figure 1. Hopewell and Bailey Bay/James River area.

Drummed residues from Kepone production were stored at te Hopewell sewage treatment plant and at Portsmouth, Virginia. Kepone-contaminated sludge from the Hopewell sewage treatment plant was stored in a lagoon constructed at the sewage treatment plant site.

Almost three years following the closure of the Life Science Products plant, the disposal of the drummed Kepone production residues and the Kepone-contaminated sludge is unresolved. Several sites in the City of Hopewell contain Kepone; small inflows of Kepone continue into the James River, and the levels of contamination remain sufficiently high in the 110 km of the river from Hopewell to the mouth to cause continued closure of the James River to recreational and commercial fishing for many species of fish and shellfish. Table 1 summarizes the estimates of Kepone residuals in the Hopewell/James River area.

TABLE 1. ESTIMATE OF KEPONE RESIDUALS

Residing In	Estimated Quantity of Kepone	
	kg	lb
Sewer System	23	50
Surface Soil (1 inch)	45 - 450	100 - 1,000
Kepone Sludge Lagoon	100	220
Bailey Bay Sediments*	540 - 2,000	1,200 - 4,300
James River Sediments*	9,000 - 17,000	20,000 - 38,000
Drums at Hopewell	9,400	20,700
Drums at Portsmouth	13,000	28,800
Landfill†	1,400	3,100
Pebbled Ammonium Nitrate Plant Site	100	220
Rounded total*	33,700 - 43,600	73,500 - 95,500

* Low value reflects estimate extrapolated from mean concentrations, high value reflects estimates based on mean plus one standard deviation.

† Includes identified deposits only.

CONVENTIONAL MITIGATION/REMOVAL APPROACHES

The project investigation of conventional mitigation/removal approaches included: (1) site surveys, analysis of world-wide dredging techniques, and assessment of the most promising dredging techniques for removal of contaminated sediments from specific sites; (2) engineering studies to contain, stabilize or remove Kepone-contaminated sediments at points of inflow into the James River, together with an assessment of their effectiveness and potential environmental impacts; and (3) evaluation of the engineering requirements for removal of Kepone-contaminated sediments from the James River, assessment of dredge spoil sites, fixation of the dredge spoil, and treatment of the elutriate.

The majority of these efforts were carried out by the Norfolk District Corps of Engineers for the Kepone Project Office and are described in detail in Appendix B of EPA's Kepone Mitigation Feasibility Report (2).

DREDGING TECHNOLOGY ASSESSMENT

The evaluation of world-wide dredging technology involved a survey of mechanical, hydraulic and pneumatic dredges. The most common dredges in the United States are the mechanical and hydraulic type. Table 2 summarizes the characteristics of the major types of mechanical and hydraulic dredges in use in the United States.

Often considerable turbidity is generated in the use of these dredges. Accordingly, when dredging contaminated sediments, such dredges pose a serious threat through secondary pollution. To control this problem, various types of silt curtains and turbidity barriers have been devised with varying degrees of success. Conventional hydraulic dredges pose additional problems in handling contaminated sediments since they collect only 10 to 30 percent solids. Disposal of dredge spoil can induce secondary pollution from runoff or leaching from the dredge spoil site, or impose high costs in treatment of the elutriate at the dredge spoil site prior to return of the liquid to the receiving water.

Dredging technology in Europe is generally similar to the capabilities in the United States. However, special purpose dredges have been devised. Of these, a pneumatic dredge, the Pneuma dredge developed in Italy, has special advantages in handling contaminated sediments. The advantages of the Pneuma include minimum wear, continuous flow, limited secondary pollution and high solids removal—60 to 80 percent solids by volume. The Pneuma dredge, however, is only practical for removal of sediments at considerable depth as the hydrostatic pressure of water is required to move the sediments into a chamber from which the material is expelled through a pipe to a containment site by compressed air.

Japanese dredging technology for handling of contaminated sediments is considerably advanced over that in the United States. Table 3 summarizes the characteristics of the major types of dredges developed in Japan. Analysis of the characteristics of these dredges and a site survey of operations in Japan indicated that the Oozer dredge does not require hydrostatic pressure to force the sediment into its chambers*. The Oozer dredge utilizes a vacuum to draw the sediment into its chambers, subsequently discharging the dredge spoil

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* Comprehensive side-by-side comparisons of potentially promising dredges operating under James River conditions do not exist. However, testing of the Oozer dredge in the James River would at a minimum provide on-site operating parameters for evaluating competing systems.

TABLE 2. MECHANICAL DREDGES IN THE UNITED STATES

Type of dredge	Method of operation	
Dragline	Contains scoop that is lowered into water and slices material to be removed as scoop is drawn toward dredge.	Scoop, shovel, or clamshell is lifted by crane and dredged material is deposited either on a barge or on the bank.
Dipper	Contains shovel that is lowered into water and slices material to be removed as shovel is drawn away from dredge.	
Grab	Contains grab or clamshell bucket. Material is removed by forcing opposing bucket edges into it. Bucket is then closed.	
Endless Chain (bucket ladder)	Includes endless chain of buckets. Material removed by forcing single cutting edge of successive buckets into material. Material deposited in barge or other conveyance.	

HYDRAULIC DREDGES IN THE UNITED STATES

Cutterhead	Rotary blades cut into bottom material. Centrifugal pump removes material with dilution water and transports it in a pipeline to disposal area.
Plain Suction	Operates same as cutterhead but without rotary cutter.
Dustpan	Material removed with water jets and picked up with dilution water by a wide but shallow suction inlet, pumped through discharge line, and returned into water adjacent to channel.
Hopper	Dredge equipped with suction pipe, draghead, and hoppers or bins that store hydraulically dredged material. There is no pipeline. Disposal is either at sea or pumped into a disposal site at the pier.
Sidecaster	Material picked up with dilution water and pumped back directly into adjacent waterway, a distance from the channel.

TABLE 3. DREDGES DEVELOPED IN JAPAN

Type of dredge	Method of operation
Clean Up	Looks like a conventional hydraulic cutterhead except for clean-up head installed on ladder. Head is equipped with a movable wing or shield to overlie bottom sediment, a movable shutter plate, an auger, a device for collecting gas, and sonar devices to indicate elevation of material before and after dredging.
Anti-turbidity	Overflow system on hopper dredges is designed to have turbid water (slurry hopper dredges mixture) that is discharged overboard quickly submerged. Discharged slurry settles rapidly to the bottom instead of remaining suspended in the water column.
Oozer	Comparable to Pneuma pump in utilizing water pressure to raise material to be dredged. In addition suction is increased by creating a vacuum in the tank. After material has filled tank, compressed air forces sludge to be discharged. Transports sludge in high density and without causing significant turbidity. Well suited for removal of viscous sediments.
Watertight Grab Bucket	Closed type. Especially designed for dredging without giving rise to secondary pollution when used for dredging settled sludge.

under air pressure directly by pipeline to a dredge spoil site or barge. This dredge is capable of removing dredge spoil with high solids content, operates at minimal draft, effectively controls secondary pollution, and has shown minimal operational difficulties under a range of sediment conditions.

ENGINEERING STUDIES TO CONTAIN CONTAMINATED INPUTS INTO THE JAMES RIVER

Engineering approaches for preventing continuing inflows of Kepone contamination into the James River were investigated. Eighteen engineering alternatives were proposed and evaluated. These alternatives considered the feasibility and utility of checking Kepone contamination from Bailey Creek and Gravelly Run which discharge into Bailey Bay, as well as removal and containment of contamination from Bailey Bay which discharges into the James River.

The alternatives were limited to structural solutions such as dredging; various types of levee, dam and wall construction; channel improvement or modification; covering or sealing; and other combinations of structural solutions. The analysis included an investigation and evaluation of the engineering feasibility, implications, and cost for removing the Kepone-contaminated sediments from Bailey Bay and Bailey Creek areas. Based on related project efforts, it was concluded that implementation of most of these alternatives was viable only if complete cleanup of the James River were contemplated. However, two of the alternatives offered potential utility and benefits as dredge spoil sites in Bailey Bay for currently contemplated maintenance dredging.

ENGINEERING STUDIES FOR REMOVAL OF KEPONE FROM THE JAMES RIVER

Both complete removal of Kepone-contaminated sediments from the James River and partial removal of contaminated sediments from "hot spots" were evaluated. Figure 2 illustrates key physical features of the area.

Complete Removal

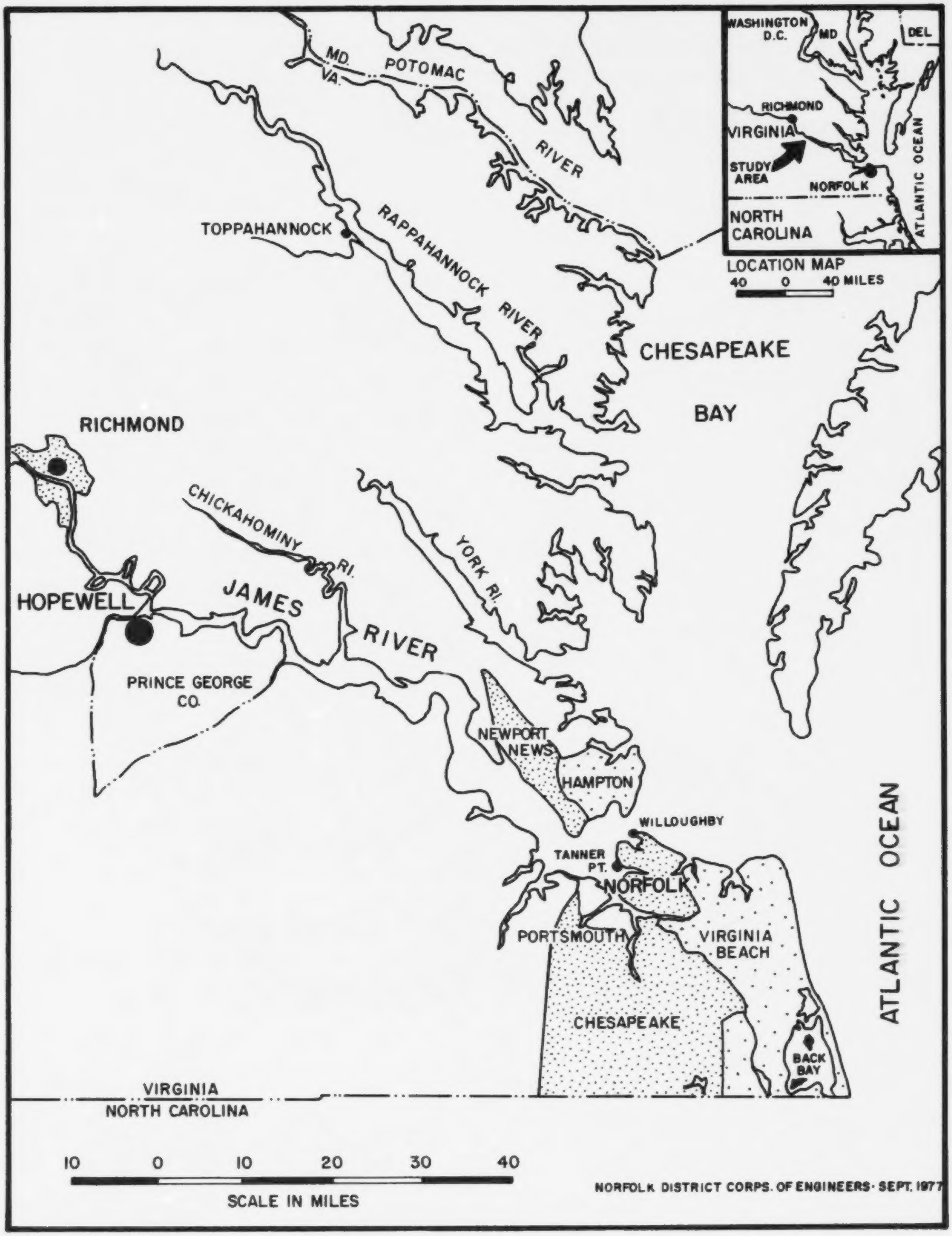
Complete removal of contaminated sediments would involve approximately 110 kilometers (69 miles) of river from Hopewell to the mouth. These sediments are generally characterized as silty clay, less than 64 microns.

The parameters and conditions considered in the analysis of full-scale removal of sediments from the James were as follows: excavation would be limited to the James River from Hopewell to the James River Bridge (no dredging was considered in tributaries of the James); excavation depth would be limited to 38 centimeters; disposal be limited to adjacent sites; sand for disposal area construction would be within an economical pumping distance of each site; the Oozer dredge would be available; approximately 25 percent excess material would be removed due to over-dredging; and the Oozer pipeline has a capability to pump material 1,524 meters (5,000 ft).

Thirteen disposal areas were proposed for confinement of Kepone contaminated spoil. Preference was given to sites contiguous to the shore and care was taken to select locations that would have minimal impact on adjacent drainage patterns. Some filling of interior low areas was anticipated. Design levels, based on 100-year flood level, dictated an elevation of 3 meters (10 ft) above sea level datum. Areas not utilized to capacity would provide dredge spoil sites for future maintenance dredging.

The results of this analysis are shown in Table 4. Removal of 133 billion cubic meters (169 billion allowing a 25 percent excess) of material and disposal would cost approximately $1 billion.

If elutriate treatment and spoil fixation costs are considered, the total cost of the project would range from $1 to $7.2 billion, depending on the treatment chosen. Elutriate treatment costs for ultra-violet/ozone were calculated considering a portable UV/oxone unit and a treatment rate of $.06/ 1,000 liters ($0.23/1,000 gallons), excluding logistics or capital costs or possible water clarification costs. A summary of a complete treatment costs estimate for treating the James River sediments with intra-basin disposal is presented in Table 5.



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RICHMOND
VIRGINIA
STUDY AREA
NORFOLK
NORTH CAROLINA
ATLANTIC OCEAN
LOCATION MAP
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MD.
VA.
POTOMAC
RIVER
TOPPAHANNOCK
RAPPAHANNOCK RIVER
CHESAPEAKE
BAY
RICHMOND
CHICKAHOMINY RI.
YORK RI.
HOPEWELL
JAMES
RIVER
PRINCE GEORGE CO.
NEWPORT NEWS
HAMPTON
WILLOUGHBY
TANNER PT.
NORFOLK
PORTSMOUTH
VIRGINIA BEACH
CHESAPEAKE
BACK BAY
ATLANTIC OCEAN
VIRGINIA
NORTH CAROLINA
10 0 10 20 30 40
SCALE IN MILES
NORFOLK DISTRICT CORPS. OF ENGINEERS · SEPT. 1977

Figure 2.

TABLE 4. SUMMARY FOR CONVENTIONAL REMOVAL OF KEPONE CONTAMINATED SEDIMENTS IN THE JAMES RIVER

Disposal Areas	Acres	Elevation of Slurry mse	Dredged Quality in Cubic Yards 15" depth	25% Excess Included	Dredging Costs at $4.30/cu yd for 15" depth	Disposal Site Preparation Cost
1	444	10.4	7,440,000	9,300,000	$31,992,000	$3,550,000
2	560	10.3	8,610,000	10,762,500	37,023,000	4,480,000
3	248	7.1	2,790,000	3,487,500	11,997,000	7,610,000
4	411	8.9	6,740,000	8,425,000	28,982,000	11,990,000
5	276	8.4	5,050,000	6,312,500	21,715,000	11,050,000
6	767	6.7	10,440,000	13,050,000	44,892,000	9,200,000
7	736	5.4	12,830,000	16,037,500	55,169,000	15,110,000
8	907	7.4	18,440,000	23,050,000	79,292,000	15,340,000
9	533	9.3	11,300,000	14,125,000	48,590,000	13,950,000
10	872	7.5	14,260,000	17,825,000	61,318,000	11,620,000
11	1572	10.4	37,780,000	47,225,000	162,454,000	17,380,000
12	1635	9.2	28,460,000	35,575,000	122,378,000	20,200,000
13	725	8.4	12,780,000	15,975,000	54,954,000	10,840,000
TOTALS	9766	ROUNDED	176,920,000 (177,000,000)	221,150,000 (211,000,000)	$760,756,000	$153,120,000
				ROUNDED	$761,976,000* (762,000,000)	$220,490,000* (220,500,000)

Total Cost for Dredging and Disposal Including Contingencies
Engineering and Design Studies and Administrative Costs

Total Project Costs = Total Dredging Cost + Total Disposal Cost

$\$902.5 \times 10^6 = \$762 \times 10^6 + \$220.5 \times 10^6$

$1 billion = Rounded

(Total removal and disposal costs amount to $5.55/cubic yard.)

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*Total cost includes contingencies, engineering and design studies and administrative costs.

TABLE 5. TREATMENT COST ESTIMATES FOR ALTERNATIVES ON THE JAMES RIVER

	Without Dredging	With Dredging
Dredging With Oozer Dredge (COE)*	N/R	$\$1.0 \times 10^{9}$
Molten Sulfur Stabilization	$\$6.2 \times 10^{9}$	$\$7.2 \times 10^{9}$
TJK Fixation with Removal	$\$1.8 - 2.6 \times 10^{9}$	$\$2.8 - 3.6 \times 10^{9}$
Elutriate Treatment - UV/ozone	12.4×10^{6}	$\$1.01 \times 10^{9}$
Elutriate Treatment - temporary scheme filtration/carbon adsorbtion	$\$40.3 \times 10^{6}$	$\$1.04 \times 10^{9}$
UV/ozone for Sediments	$\$26.6 - 53.1 \times 10^{6}$	$\$1.03 - 1.05 \times 10^{9}$
In Situ (Battelle)*		
Application of Retrievable Sorbents	$\$6.2 \times 10^{9}$	N/R
Application of Coal	$\$2.2 \times 10^{8}$	N/R
Application of Activated Carbon	$\$3.6 \times 10^{9}$	N/R

N/R - Not required

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* The areas used by the Corps of Engineers (COE) for determining dredging alternative costs were slightly different than those used by Battelle in determining non-conventional alternative costs. The relative ranking of alternatives due to cost determinations remains unchanged even with this difference.

Battelle Pacific Northwest Laboratories, in separate efforts, determined costs and some environmental consequences for other in situ mitigation proposals. These results are also included in Table 5.

MITIGATION THROUGH CLEANUP OF ELEVATED AREAS OF CONTAMINATION

Certain areas of the James River contain more Kepone than others as a result of the discharge patterns and hydrologic influences. For example, Bailey Bay sediments contain up to 10 μg/g (ppm) of Kepone as a result of proximity to the discharge source at Hopewell. Concentrations of the order of 1 μg/g (ppm) of Kepone are contained in the sediments of the turbidity maximum zone as a result of estuarine hydrologic influences. The Battelle model was utilized to evaluate the effectiveness of partial clean-up.

Ten selected zones of high Kepone sediment contamination were designated. The local effects of cleanup were assessed as well as the effect near the mouth of the James River discharging into Chesapeake Bay. Results were compared with the no-cleanup case for total, dissolved and suspended particulate Kepone (attached to sediments). The model results indicated essentially no improvement locally or near the mouth of the James River through cleanup of Bailey Bay. Only two cases of cleanup involving 32 to 56 kilometers (20 to 35 miles) of river reach in the turbidity maximum zone showed significant localized improvement. None of the cleanup options has any appreciable effect on the contamination near the mouth of the James River. In addition, none of the partial cleanup options reduced the levels of contamination sufficiently to eliminate the hazard of bioaccumulation of Kepone by fish to the action levels. It was therefore concluded that "hot spot" or partial cleanup operations could not be recommended. However, partial cleanup of "hot spots" would potentially reduce the time for closure of the river due to contamination. Accordingly it was recommended that the dredge spoil from maintenance dredging of the river be removed to safe land containment sites.

NONCONVENTIONAL MITIGATION APPROACHES

In addressing nonconventional mitigation methods, the project team focused on evaluating alternatives to dredging, as well as treatment and/or fixation processes complementary to dredging for application to Kepone contaminated sediments in the James River System. Four types of alternative approaches were studied: dredge spoil fixation, elutriate treatment, in situ stabilization, and isolation.

DREDGE SPOIL FIXATION

Dredge spoil fixation is necessary to prevent leachate of contaminated materials from the dredged spoil site. Fixation processes investigated were classified according to base ingredients. These included: silicates, organics, sulfur, gypsum, and asphalt. Twenty-one samples were tested and the sampling included eight different companies. A silicate-based, organic-based and sulfur-based fixation agent showed the most promise for the Kepone application.

Silicate-Based Agent

In general, most silicate-based agents rely on high pH conditions to set the stabilized material. Kepone is solubilized under these conditions and if the fixation additive does not isolate the Kepone, it will occur in leachate at equivalent or higher levels than is found with natural sediments. The Japanese firm Takenaka utilizes a silicate-based agent. However, they improved their additive agents over the first series of tests which indicated no better fixation capabilities than natural sediments. Results of the latest fixation efforts on Bailey Bay sediment samples showed leachate Kepone concentrations of only 0.08 μg/l (ppb) in fixed sediments which originally showed a leachate of .57 μg/l (ppb) of Kepone. Takenaka officials believe that they can reach a level of 0.01 μg/l (ppb) to 0.03 μg/l (ppb).

The Takenaka technology offers the advantage over other fixation processes in that fixation can be performed in-place. Most processes necessitate removal, treatment and then replacement of the fixed material, thus adding considerably to the costs of the process. In addition, the Takenaka technology has had widespread operational use in Japan under a range of conditions. Large scale projects have been ongoing for several years in contaminated harbors and rivers in Japan, and involve fixation of contaminated industrial sludge, contaminated dredge spoil, and *in situ* fixation operations on contaminated bottom sediments at depths up to 40 meters. To date, the fixation processes have been used effectively on sludge contaminated by mercury, copper, zinc, cadmium, lead, chromium, and PCBs. Laboratory tests have shown the processes to be effective on arsenic as well.

Organic-Based Agent

Por Rok Epoxy Sealant, an organic base fixation agent, is a synthetic epoxy material which is mixed with a coarse aggregate material and is used as a grout or surface sealant. It reduced Kepone leachate concentrations an order of magnitude lower than those of standard sediments. This stabilization agent showed promise as a means of reducing Kepone releases from spoils. However, application costs are high - \$442/m^3 (\$12.50/ft^3) and the material is limited in availability.

Sulfur-Based Agent

Molten sulfur used as a fixation agent offered an order of magnitude reduction of leachate concentrations in laboratory tests. However, the practicality of molten sulfur for a large scale application to Kepone contaminants is unknown. Furthermore, it is recognized that there could be severe evironmental impacts associated with this dredged spoil fixation process because elemental sulfur, while stable in water, readily changes to soluble and potentially toxic forms when mixed with reducing as well as oxidizing sediments. Molecular compounds of concern include carbon disulfide, hydrogen sulfide and sulfur dioxide. These compounds should be handled carefully. Accordingly, the molten sulfur technique will require additional investigation and evaluation.

ELUTRIATE TREATMENT

The elutriate associated with contaminated dredged spoil material must be treated prior to return of the elutriate to the receiving waters. This treatment is especially critical with Kepone-contaminated elutriate since Kepone in low concentrations can be bioaccumulated rapidly.

Elutriate/slurry treatment methods investigated consisted of: photochemical degradation; amine photosensitization; chlorine dioxide, ozone and UV/ozone treatment in combination radiation with gamma rays and electron beams; catalytic reduction; and carbon adsorption.

Based on the study investigations, the UV/ozone and the temporary filtration/carbon adsorption scheme were deemed best suited for elutriate treatment.

Ultra-Violet/Ozone Treatment

Research on combined ozonation and ultra-violet irradiation by Westgate Research Corporation exhibited better than 80 percent Kepone removal from the effluent of Hopewell's primary treatment plant when samples were subjected to exposure periods of one hour.

Cost projections for large treatment plants, developed by Westgate Research for other applications indicate a capital cost of $33,022 to $37,000 per MLD ($125,000 to $140,000 per MGD) capacity and operation and maintenance costs, including amortization, of $.03 per 1,000 liters ($0.11 per 1,000 gallons) treated.

In continued research to determine whether ultraviolet irradiation processes are limited to clear samples receiving direct irradation, Westgate treated sediment slurry samples taken from a contaminated creek bed. There was a significant loss of Kepone, 63.8 percent, in the sediment during the first 30 minutes of reaction time. It is believed that Kepone is being destroyed in the water phase, and that the partition coefficient permits continuous release of Kepone from the sediment to the water. This could explain the relatively constant values obtained in the supernatant analyses. The sediment, about 20 percent by volume, was held in suspension in the "Ultrox" reactor by the ozone spargings (Westgate, 1978). This ability to handle high solid content slurries also holds promise for the direct treatment of dredged slurries or sediments, if the sediments are put in slurries of 20 to 50 percent solids. However, further evaluation of the process is required to determine the extent of Kepone degradation and assess the potential toxicity of the by-products. It appears that degradation occurs by the removal of chlorine from the Kepone molecule forming monohydroKepone.

From these very preliminary but encouraging results, it is estimated that an optimum large-scale portable treatment system could treat 20 to 50 percent solids slurry for 13 to 26 cents per cubic meter (10 to 20 cents per cubic yard), not including equipment amortization. However, further testing is needed to adequately define operating parameters and associated costs.

No major environmental impacts are anticipated with the application of the UV/ozone treatment process other than those associated with construction of the facilities and the increased demand for power. However, as indicated, by-products and/or deleterious residues have not been studied.

Filtration/Carbon Adsorption

During the decontamination efforts in early 1976, the EPA mobile spill treatment unit was brought to Hopewell to help decontaminate washwaters and liquid wastes. Carbon adsorption proved effective in removing Kepone from solution. Accordingly, the Kepone project team contacted companies experienced in large scale applications of carbon adsorption treatment.

The Calgon Corporation has developed a filtration/carbon adsorption wastewater treatment system which shows promise in treating dredge slurry water containing Kepone. In their conceptual design, the dredged slurry would be pumped from the dredge to an impoundment basin where the solids would settle from the water. From the impoundment basin the water would flow to a gravity slow sand filter, constructed in a lined earthen basin. From the sand filter the water would flow by gravity to an adsorption basin and be directed through a bed containing a layer of gravel and a layer of Filtrasorb activated carbon. From the filter bed, operated in a flooded condition to prevent channeling, the treated water would overflow to a spillway and be returned to the river. At the conclusion of the dredging, the entire sand filtration and carbon beds could be encapsulated and backfilled to prevent future leachate contamination. Calgon estimates the capital costs for a 50 MLD unit at $.81 million (50 MGD unit at $3.06 million). More extensive analysis of the site and operational requirements will be required to develop operating costs.

Comparison of the Ultra-Violet Light/Ozone and Filtration/Carbon Adsorption

Both the UV/ozone treatment system, proposed by Westgate Research Corporation, and the temporary filtration/carbon adsorption wastewater treatment system proposed by Calgon Corporation were recommended for further evaluation for elutriate treatment. The UV/ozone option seems to destroy Kepone, but the degradation products and their relative toxicity still need to be determined. The use of the temporary filtration/carbon adsorption option may still pose disposal problems if future leachate contamination is to be prevented.

IN SITU STABILIZATION AND ISOLATION

In situ stabilization processes as a category are the newest of the approaches to removal/mitigation of in-place toxic materials. As such, most are less developed than other approaches. Several of the more promising new options were selected for testing in the Battelle laboratory. Since biological approaches currently evaluated appear to offer little with respect to the removal of Kepone from the James River system, work focused on two types of *in situ* approaches: use of sorbents and use of polymer films. In addition to these approaches, the Japanese Takenaka fixation process previously described can be used in *in situ* mitigation of contaminants. (The Japanese fixation process is a proven large scale operational in-place fixation technology. This fixation process generally costs $13 to $20 a cubic meter, and eliminates removal costs). However, indications are that the top few centimeters at the sediment/water interface may be difficult to fix. Consequently, such an application for Kepone, with the present state of knowledge, is far less desirable than removal of the contaminated sediments and fixation in carefully contained dredge spoil sites. Also any in-place fixation technique may have major impacts on benthic communities.

Natural sorbents (such as activated carbon) and synthetic sorbents (such as the macroreticular resins) have been shown to be effective in concentrating organics similar to Kepone. In application, sorbents act much as natural sediments do in maintaining levels of Kepone much higher than those in adjacent waters. Sorbents having lower partition values (concentration in water/ concentration in substrate) than those exhibited in natural sediments will reduce the levels of dissolved Kepone in the water if introduced to the system. A three-phase equilibrium is established with the highest concentrations of Kepone on the new material, a lower concentration on the sediment, and the lowest concentration in the water.

Based on Battelle's initial screening results sorbents ES863, XAD-4, XAD-2, and Filtrasorb 300 were selected for further study. The three proprietary products are macroreticular synthetic sorbents produced commercially. Filtrasorb 300 is a commercial activated carbon. In addition to these, a specialty carbon product formed around iron particles became available in time for subsequent evaluation. Allied Chemical also performed laboratory tests utilizing anthracite coal.

Following Allied Chemical Company's promising initial results with coal, batch adsorption tests were initiated in the Battelle laboratory on a variety of coals. Results indicate that coals tested had less affinity for Kepone than Bailey Bay sediments. Consequently, these coals did not offer any mitigation utility for Bailey Bay sediments. However, Bailey Bay sediments are high in organic content and testing on more representative James River sediments should be undertaken before final determinations are made on the applicability of coal.

Although sorbents applied to sediments *in situ* are capable of reducing the availability of a material to the water column, they do not destroy or remove the contaminant. Removal can be achieved, however, if media are made to be retrievable. Laboratory work at Battelle indicates that this is possible through the inclusion of magnetite or iron particles in the sorbent matrix which will render the media particles susceptible to magnetic fields. However, the practical application is unevaluated. The magnetic sorbents would have to be mixed into the river sediments and then recovered. Strong magnetic fields may be required and dispersion of contaminants avoided. Costs for using synthetic sorbents are prohibitively expensive on wide scale application unless they can be made retrievable. Even then, costs of using synthetic sorbents is estimated to be \$31/m^3, (0.90/ft^3) sediment compared to \$1.1/m^3 (\$0.032/ft^3) sediment for coal and \$18/m^3 (\$0.52/ft^3) sediment for activated carbon.

It was noted previously that activated carbon had been successfully applied to remove Kepone from solution. The same is true for any of the sorbents found to be effective for *in situ* evaluations such as XAD-2 and ES863. Consequently, sorbents, if developed for operational application, should be considered as candidates for both elutriate and *in situ* application. In *in situ* use, activated carbon might be considered for application directly to the river without retrieval. This would be done in the same manner as an application of coal, but activated carbon offers a high degree of adsorption. However, availability of the resultant contaminated carbon to the biota has not been evaluated.

Of the *in situ* approaches considered, all show some degree of effectiveness. The potential of using coal is still not resolved. Based on laboratory comparisons, activated carbon is more appropriate than coal as an *in situ* additive. Any *in situ* use of activated carbon or coal, if it proves to be operationally effective, would be limited in application to areas contaminated to less than 1 μl/g (ppm). In areas where Kepone concentrations were greater than or equal to 1 μg/g (ppm), retrievable media or fixation techniques should be given further consideration. The latter exception is made to reflect the fact that at high Kepone concentrations in sediments, the potential reduction in Kepone availability with coal or activated carbon would still allow unacceptable levels of Kepone in the water. For these areas, retrievable media and fixation, if effective, would be costly.

Environmental impacts associated with *in situ* treatment are not well understood. However, *in situ* treatments potentially pose many physical, biological and chemical impacts on the river and its biota. Accordingly, no *in situ* treatment approaches are recommended at this time for the James River.

NATURAL DEGRADATION AND BIOLOGICAL MITIGATION APPROACHES

As part of the Kepone Mitigation Feasibility Project, natural degradation and biological approaches seemed viable avenues for amelioration or elimination of the Kepone contamination problem. They were examined concurrently with engineering approaches. These methods, if they proved successful, could offer the possiblity of in-place mitigation without elaborate or expensive engineering programs or structures. Our investigation of potential biological mitigation methods included: (1) degradation or destruction of Kepone by microorganisms; and (2) biological uptake and concentration of Kepone by an animal or plant with subsequent harvest and destruction. Natural biodegradation of Kepone was the most hopeful route because many substances are capable of some degredation by natural means.

DEGRADATION

A search of the literature indicated only indirect and circumstantial evidence that Kepone could degrade, either photochemically or biologically. Laboratory efforts were aimed at degradation studies under a variety of conditions, including aerobic and anaerobic situations. The EPA Gulf Breeze Environmental Research Laboratory and Battelle Pacific Northwest Laboratories performed experiments to determine if Kepone degradation would occur under laboratory conditions.

Studies by Garnas et al. (3) and Bourquin et al. (4) of Gulf Breeze employed static water/sediment systems to assess both biological and nonbiological degradation of Kepone. James River sediments with and without Kepone were used and the fate of Kepone monitored with radio-labelled Carbon 14 material and with total budget chemical analyses. The investigators employed a variety of experimental conditions including oxygen concentration, nutrient additions, Kepone levels, sediment sources, sunlight, temperature, and salinity.

Gulf Breeze experiments indicated that Kepone does not degrade either biologically or chemically in the laboratory situations, which simulated natural states as closely as possible. These results indicate that currently evaluated degradation processes will not reduce the levels of Kepone now found in the water and sediment of the James River.

The Battelle Laboratories found results similar to those of Gulf Breeze. While some evidence exists that certain bacteria can survive in the presence of Kepone-contaminated sediments, degradation of Kepone has not been demonstrated or confirmed. The half-life of Kepone in the environment has not been determined, but laboratory evidence suggests it may be on the order of decades.

Fungi offered a potential for Kepone degradation, and Atlantic Research Corporation believes they have a series of fungi which will degrade Kepone. One isolate showed 41 percent disappearance of Kepone in 22 to 31 days. However, these experiments have been performed only in the laboratory, and no scaling-up has been attempted. The fungi may have difficulty competing and surviving under natural conditions, thus their usefulness is restricted to controlled environments, such as containment lagoons. Such an approach would not be feasible for the amounts of Kepone-contaminated sediments in the James River which need treatment.

BIOLOGICAL UPTAKE AND HARVESTING

Studies with other chlorinated hydrocarbons have shown that they can be bioconcentrated by plants and that uptake increases with water solubility. However, studies to date with Kepone indicate that plant uptake is not an effective mechanism for mitigating Kepone in the environment.

Algae were found to bioaccumulate Kepone to as much as 800 times the ambient levels of Kepone (5). This approach was examined for the possibility that Kepone-contaminated algae could be harvested and destroyed by incineration to eliminate the Kepone. Problems, similar to those of fungi, became apparent. Furthermore, the algae bioconcentrated Kepone from the water instead of the sediments where the bulk of the Kepone is located.

In a similar fashion, Kepone attached to rooted aquatic plants such as water hyacinths, could be isolated, harvested and destroyed. Although the water hyacinths may accumulate Kepone on their leaf surfaces, the plants are free-floating and would not be in contact with the large percentage of Kepone which resides in the James River sediments, giving this method limited value for mitigation purposes.

Although Kepone is intimately bound to sediments of the James River, it can desorb from those sediments, or organisms can extract it from the sediments to become incorporated into living systems. It can be passed between organisms readily as one animal uses another as food. Kepone is also available to animals from the water. Although animals can bioconcentrate Kepone, they were never viewed to have utility for Kepone uptake with subsequent harvesting to destroy the Kepone. As with plant harvesting the logistics are impractical. Ultimately any mitigation method proposed to ameliorate the effect of Kepone must address the bioconcentration of Kepone by James River organisms, which traditionally have been used as a food resource.

SUMMARY OF RECOMMENDED MITIGATION APPROACHES

The project findings which lead to the recommended mitigation approaches are too extensive to detail here. These are covered in the Kepone Mitigation Project Report (2). However, some of the key findings will be summarized here as they relate to specific recommended mitigation approaches. A special task force consisting of representatives from EPA and the States of Virginia and Maryland has already been established to consider and implement the recommendations.

CHESAPEAKE BAY MITIGATION APPROACHES

Modeling and monitoring studies indicated that Chesapeake Bay is not currently threatened by the contamination of the James River. However, severe storms and like events could alter the situation. Accordingly, continued systematic monitoring of Kepone levels in water sediment and biota was recommended. In addition, development of a long range strategy including use of submerged silt dams to implement emergency mitigation measures was recommended.

JAMES RIVER MITIGATION APPROACHES

Based on the enormous costs of total James River amelioration efforts, the lack of knowledge on ecological impacts of widespread mitigation efforts, the unavailability of economic impact determinations, and supportive evidence that most of the Kepone will remain in the zone of turbidity maximum, no full-scale cleanup action was recommended for the James River at this time. However, since navigational dredging is required in the James River, these dredging operations afford a continuing opportunity to initiate removal of the contaminated sediments. Accordingly, it was recommended that the Oozer dredge be considered for such dredging and the dredge spoil be disposed of in adequately protected dredge spoil sites developed along the James River.

Among the key issues deterring progress in physical removal of the contaminated sediments from the James River were uncertainties regarding the effectiveness and costs associated with treatment of elutriate and dredge spoil. However, promising technologies were found whose further development was recommended. Two of the technologies were recommended for developmental funding—the Takenaka fixation technology developed for Kepone-contaminated sediments and the UV/ozone treatment of Westgate Research.

HOPEWELL AREA MITIGATION APPROACHES

Contamination of the land around the Hopewell area is extensive. Some limited land areas within the city limits, accessible to the population, have levels ranging to 1,530 μgg (ppm). Accordingly, it was recommended that the health threat be assessed—the most expeditious action being removal of the contaminated soil and disposal.

A marsh area contiguous with a creek discharging into Bailey Bay and thence the James River was found to contain an estimated 1,363 kg (3,000 lb) of Kepone. Containment or physical removal of the contaminated material was recommended.

A lagoon containing Kepone-contaminated sludge obtained from the early cleanup operations around Hopewell has suspected leakage into a nearby creek. It was recommended that the material be removed from the lagoon or covered and containment be initiated.

CLOSURE

United States' experience in handling in-place toxic pollutant problems of the extent exhibited by the Kepone contamination in the Hopewell/James River area is limited. However, adaptation of Japanese technology in dredging and sediment fixation holds promise for substantial assistance in solving such problems. In addition, it is hoped that the investigative techniques and findings of the Kepone Mitigation Feasibility Project will not only provide a basis for moving forward with the solution of the Kepone contamination problem, but will provide an approach to solution of other serious waterway contamination problems in the United States.

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PCB CONTAMINATION OF THE SHEBOYGAN RIVER, INDIANA HARBOR AND SAGINAW RIVER AND BAY

Karl E. Bremer
U.S. Environmental Protection Agency
Chicago, Illinois 60604

ABSTRACT

Current PCB contamination in three harbor areas of the Great Lakes has resulted in problems related to maintenance dredging, disposal, or fish contamination. The Sheboygan River shows high PCB residues in fish and bottom sediments as a result of improper PCB disposal. Indiana Harbor, one of the most highly contaminated harbors in the Great Lakes is experiencing delays in maintenance dredging of approximately 750,000 cubic yards of bottom sediment. High concentrations of mercury, lead, zinc, manganese, arsenic, and PCBs have been detected throughout the harbor and ship canal. A new dike disposal area has been completed in Saginaw Bay. Possible use of this confined disposal site will be made for PCB-contaminated sediments from the Saginaw River.

INTRODUCTION

There are approximately 180 harbor areas in the Great Lakes Basin which are served by commercial shipping. To maintain a minimum depth of 8.2 meters (27 feet) in these harbors, continued maintenance dredging occurs at various harbor areas along the extensive shoreline of the Great Lakes.

The United States and Canada continue to maintain these ports with a joint determination to restore and enhance water quality. In the Agreement on Great Lakes Water Quality of April 15, 1972, the Governments of the United States and Canada directed that specific attention be given to assessing the potential for deleterious environmental aspects from dredging activities within the Great Lakes basin. In addition, two Acts have been passed within the last three years to specifically deal with toxic substances problems in both countries. In Canada, the Environmental Contaminants Act, and in the United States, the Toxic Substances Control Act have been implemented. These Acts will definitely impact dredging activities on the Great Lakes during the next few years.

As part of the U.S. role in maintaining Great Lakes water quality, toxic contaminant problems were evaluated in three waterways of particular importance: the Sheboygan River, Indiana Harbor, and Saginaw River and Bay.

SHEBOYGAN RIVER

The Sheboygan River and its tributaries, the Mullet River, Onion River, Weedens Creek, and Greendale Creek flow east to the western side of Lake Michigan in the State of Wisconsin. The River enters Lake Michigan through a 450-foot-wide outer basin of Sheboygan Harbor.

During 1975, 1976 and 1977, the Wisconsin Department of Natural Resources (the State pollution control agency in Wisconsin) analyzed Lake Michigan and Sheboygan Harbor fish for PCB contamination (1). Average PCB concentrations in Sheboygan Harbor fish were similar to PCB levels found in Lake Michigan fish along the coast (Table 1). Average PCB concentrations in Sheboygan Harbor sediments in 1977, as analyzed by the U.S. Environmental Protection Agency (Table 2) were also low when compared with other Lake Michigan harbors (2).

In March 1978, the Wisconsin Department of Natural Resources analyzed fish from the Sheboygan River at a location three miles upstream from Sheboygan Harbor. Test results for whole fish ranged from 26 to 750 μg/g (Table 3). To confirm these analyses, additional samples were obtained March 31, 1978. These results ranged from 8.3 to 241.5 μg/g. PCB concentrations in all fish exceeded the U.S. Food and Drug Administration's temporary tolerance level of 5 μg/g.

In April 1978, the Wisconsin Department of Natural Resources and Department of Health and Social Services advised the residents of Wisconsin not to eat fish from the Sheboygan, Mullet and Onion Rivers. During the same period of time, an investigation was conducted to test fish, river sediments, municipal and industrial effluents, and river water in the Sheboygan River drainage basin.

Fish were collected for PCB analysis at 11 locations in the Sheboygan River drainage basin (Figure 1). PCB concentrations in fish taken from the Sheboygan River lakeward from Sheboygan Falls Dam to Lake Michigan ranged from 0.9 to 970.0 μg/g (Table 4). The mean concentration for these samples was 155.0 μg/g. Thirty-eight fish samples upstream of Sheboygan Falls Dam on the Sheboygan, Mullet and Onion Rivers contained PCB concentrations below the U.S. Food and Drug Administration's temporary tolerance level of 5 μg/g.

Bottom sediments were analyzed at 13 locations in the Sheboygan River drainage basin. Concentrations in sediments ranged from 0.1 to 190 μg/g PCB (Table 5). The highest PCB concentrations were detected in sediments immediately downstream from the Tecumseh Products Diecasting Plant in Sheboygan Falls. PCB concentrations showed a decline at locations further downstream from the diecasting facility.

The Wisconsin Department of Natural Resources analyzed PCBs in municipal and industrial effluents collected from facilities in the drainage basin (Table 6). In addition, PCBs were analyzed in river water and in hydraulic fluid taken from three aluminum die-casting plants. Significant PCB point sources to the Sheboygan River were not detected by monitoring these municipal and industrial discharges.

TABLE 1. PCB ANALYSIS ON LAKE MICHIGAN FISH COLLECTED OFFSHORE FROM SHEBOYGAN.

Lake Michigan Samples										Length	ppm PCB	% fat
290	60	Lk Mich	Sheboygan	1502	05/14/75	Hyg*	70169	Brown Trout	F	13.8	.003	08.0
290	60	Lk Mich	Sheb Beach	5TH	07/24/75	Hyg	56367	Brown Trout	F	19.6	05.1	12.2
290	60	Lk Mich	Sheboygan	1502	05/15/75	Hyg	70168	Brown Trout	F	24.4	03.4	015
290	60	Lk Mich	Sheboygan	1502	05/14/75	Hyg	70159	Lake Trout	F	12.2	0.50	01.1
290	60	Lk Mich	Sheboygan	1502	05/14/75	Hyg	70160	Lake Trout	F	16.7	02.4	04.3
290	60	Lk Mich	Sheboygan	1502	05/14/75	Hyg	70146	Lake Trout	F	19.5	01.8	06.0
290	60	Lk Mich	Sheboygan	1502	05/14/75	Hyg	70147	Lake Trout	F	20.0	0.09	05.3
290	60	Lk Mich	Sheboygan	1502	05/25/75	Hyg	70165	Lake Trout	F	20.4	09.0	12.1
290	60	Lk Mich	Sheboygan	1502	05/15/75	AG**	13944	Lake Trout	F	21.8	04.4	10.0
290	60	Lk Mich	Sheboygan	1502	05/15/75	AG	13945	Lake Trout	F	25.9	10.6	19.4
290	60	Lk Mich	Sheboygan	1502	05/14/75	AG	13946	Lake Trout	F	28.7	18.2	21.0
290	60	Lk Mich	Sheb Beach		07/24/75	Hyg	56368	Carp	F	23.8	02.8	09.5
290	60	Lk Mich	Sheb Beach	NTH	07/22/75	Hyg	57166	Carp	F	24.2	28.0	08.0
290	60	Lk Mich	Sheb Beach	NTH	07/22/75	Hyg	57170	Cart	F	30.5	12.0	22.0
290	60	Lk Mich	Sheboygan	1503	05/14/75	Hyg	70154	Bloater Chub	F	10.7	0.25	04.5
								AVERAGE			6.57	10.6
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05533	5 Alewifes	EP		03.4	04.0
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05534	3 W Sucker	F	14.7	22.0	02.4
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05536	3 Whitefish	F	17.0	07.7	05.4
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05535	Brook Trout	F	12.5	07.8	04.8
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05540	R Trout	F	17.8	05.4	10.1
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05542	2 Br Trout	F	18.3	06.3	12.6
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05543	2 Br Trout	F	19.7	05.1	15.3
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05541	Brown Trout	F	29.0	12.0	12.0
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05539	Lake Trout	F	22.7	07.9	12.2
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05538	Lake Trout	F	31.0	25.0	12.1
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05532	Coho Salmon	F	18.4	26.0	05.7
290	60	Lk Mich	Sheboygan	Harb	06/24/76	Hyg	05537	Chinook	F	34.5	11.0	02.6
								AVERAGE			11.6	8.3
290		Lk Mich	10 MI SE Sheb		04/19/77	Hyg	67539	5 Chubs	WF	08.7	02.0	04.9
290		Lk Mich	10 MI SE Sheb		04/19/77	Hyg	67538	5 Chubs	WF	09.5	01.8	07.3
290		Lk Mich	10 MI SE Sheb		04/19/77	Hyg	67536	5 Chubs	WF	10.0	01.5	11.5
290		Lk Mich	10 MI SE Sheb		04/19/77	Hyg	67535	5 Chubs	WF	10.3	02.6	13.3
290		Lk Mich	10 MI SE Sheb		04/19/77	Hyg	67534	5 Chubs	WF	11.1	01.7	15.5
290		Lk Mich	10 MI SE Sheb		04/19/77	Hyg	67537	5 Chubs	WF	12.0	02.3	13.3
								AVERAGE			1.98	10.9

*State Laboratory of Hygiene

**State Department of Agriculture Laboratory

TABLE 2. BULK SEDIMENT CHEMISTRY PCB AND PESTICIDES ANALYSIS
(All values are mg/kg dry weight)

HARBOR: Sheboygan small boat harbor, Wisconsin
SAMPLED: September 16, 1977

	Sample Site				
Compound	SHB77-1	SHB77-2	SHB77-3	SHB77-3 Replicate	SHB77-4
Hexachlorobenzene	*	*	*	*	*
beta Benzenehexachloride	*	*	*	*	*
Lindane	*	*	*	*	*
Treflan	*	*	*	*	*
Aldrin	<0.02	<0.02	<0.02	<0.02	<0.02
Isodrin	<0.02	<0.02	<0.02	<0.02	<0.02
Heptachlor Epoxide	*	*	*	*	*
gamma Chlordane	<0.02	<0.02	<0.02	<0.02	<0.02
o,p -DDE	*	*	*	*	*
p,p'-DDE	<0.02	<0.02	<0.02	<0.02	<0.02
o,p -DDD	*	*	*	*	*
o,p -DDT	<0.03	<0.03	<0.03	<0.03	<0.03
p,p'-DDD	<0.02	<0.02	<0.02	<0.02	<0.02
p,p'-DDT	<0.04	<0.04	<0.04	<0.04	<0.04
Methoxychlor	*	*	*	*	*
Mirex	*	*	*	*	*
2,4-D, Isopropyl Ester	*	*	*	*	*
Endosulfan I	*	*	*	*	*
Dieldrin	*	*	*	*	*
Endrin	*	*	*	*	*
Endosulfan II	*	*	*	*	*
DCPA	*	*	*	*	*
Tetradifon	*	*	*	*	*
Aroclor 1016 (1242)	0.149	0.047	0.028	0.040	0.070
Aroclor 1248	*	*	*	*	*
Aroclor 1254	0.167	0.044	0.027	0.055	0.035
Aroclor 1260	*	*	*	*	*
Total PCB	0.316	0.091	0.055	0.095	0.105

*Concentration less than 0.01 mg/kg.

TABLE 3. PCB CONCENTRATIONS OF WHOLE FISH OBTAINED FROM THE SHEBOYGAN RIVER THREE MILES UPSTREAM FROM SHEBOYGAN HARBOR

Sampling Date	Species & Length	PCB Concentration µg/g (ppm)*
September 9, 1977	5 Carp (12-23 inches)	750
September 9, 1977	5 Northern Pike (18-25 inches)	55
September 9, 1977	5 White Sucker (10-14.5 inches)	26
March 31, 1978	17 Black Bullheads (3.5-9 inches)	41.8
March 31, 1978	12 Black Bullheads (4-9 inches)	34.3
March 31, 1978	1 Rock Bass (8 inches)	10.0
March 31, 1978	2 Common Shiners (4 inches)	49.8
March 31, 1978	1 Coho Salmon (23 inches)	8.3
March 31, 1978	2 Walleye (17 inches)	241.5
March 31, 1978	3 Carp (10-18.5 inches)	180.0
March 31, 1978	4 Carp (23-24 inches)	158.4
March 31, 1978	5 Suckers (12-21 inches)	30.0
March 31, 1978	4 Northern Pike (21-27 inches)	62.6

*ppm - parts per million

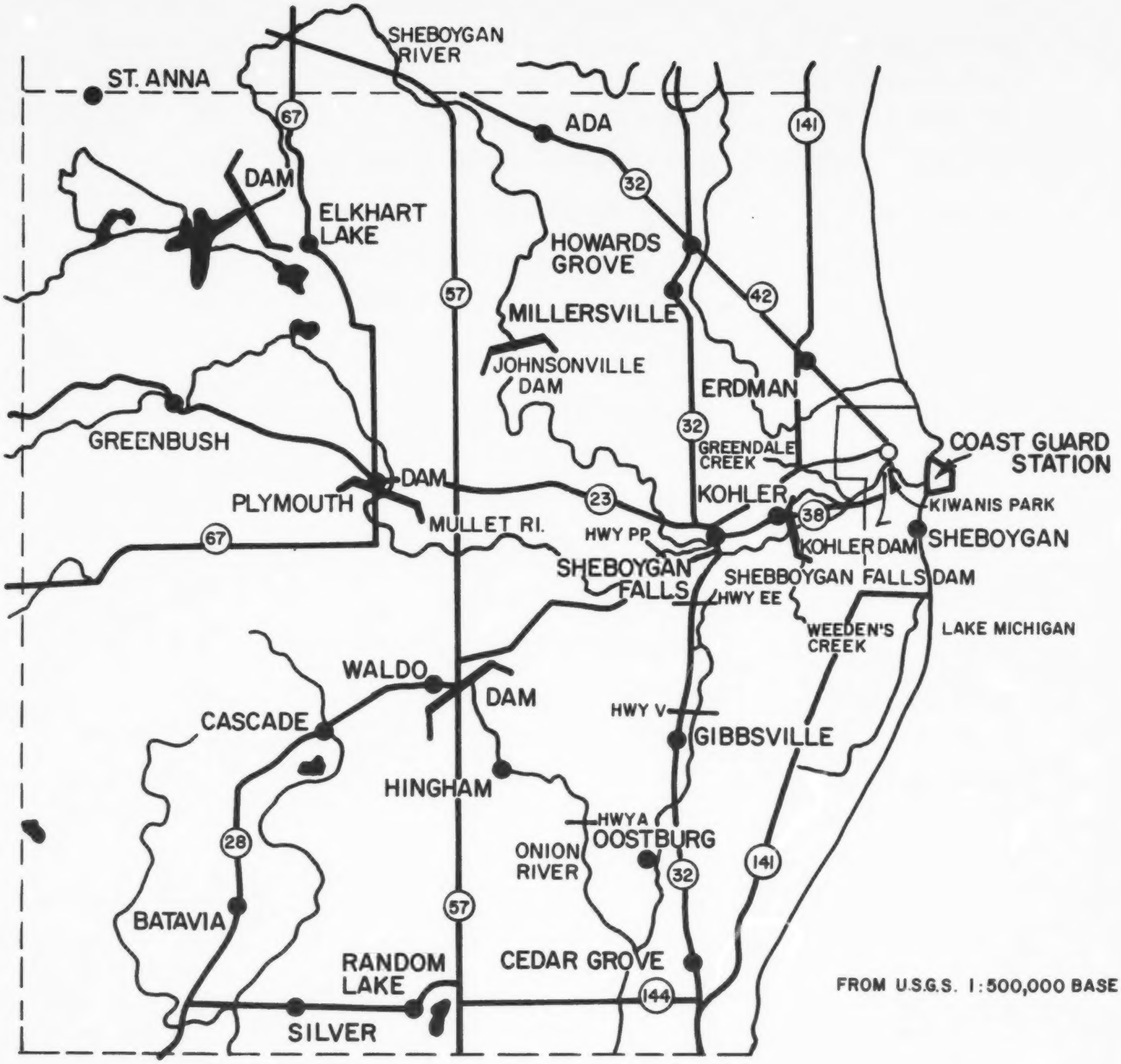
SHEBOYGAN RIVER
ST. ANNA
67
ADA
141
DAM
32
ELKHART LAKE
HOWARDS GROVE
57
42
MILLERSVILLE
JOHNSONVILLE DAM
ERDMAN
GREENBUSH
32
GREENDALE CREEK
COAST GUARD STATION
DAM
PLYMOUTH
23
KOHLER
KIWANIS PARK
MULLET RI.
38
67
HWY PP
KOHLER DAM
SHEBOYGAN
SHEBOYGAN FALLS
SHEBBOYGAN FALLS DAM
HWY EE
WEEDEN'S CREEK
LAKE MICHIGAN
WALDO
DAM
HWY V
CASCADE
GIBBSVILLE
HINGHAM
HWY A
28
ONION RIVER
OOSTBURG
32
141
57
BATAVIA
RANDOM LAKE
CEDAR GROVE
144
SILVER
FROM U.S.G.S. 1:500,000 BASE

Figure 1. Sheboygan River drainage basin (PCB investigation area).

TABLE 4. PCB DATA ON SHEBOYGAN RIVER FISH SAMPLES

Sampling Date	Species and Average Length	Sampling Location	PCB Concentration µg/g (ppm)***
Kiwanis Park Samples			
5/16/78	1 Carp (27 in.)*	Kiwanis Park-Sheboygan	15.3
5/16/78	1 Carp (23 in.)*	Kiwanis Park-Sheboygan	17.0
5/16/78	1 Carp (22 in.)*	Kiwanis Park-Sheboygan	12.1
5/16/78	1 Carp (25 in.)	Kiwanis Park-Sheboygan	2.0
5/16/78	1 Carp (23.5 in.)*	Kiwanis Park-Sheboygan	333.0
5/16/78	1 Carp (24.5 in.)*	Kiwanis Park-Sheboygan	970.0
5/16/78	4 Suckers (14 in.)**	Kiwanis Park-Sheboygan	23.9
Greendale & Weeden's Creek Samples			
4/28/78	1 Northern Pike (15.0 in.)**	Greendale Creek	169
4/28/78	8 Minnows (6.0 in.)**	Greendale Creek	5.9
4/28/78	17 Minnows (4.5 in.)**	Greendale Creek	14.0
4/28/78	1 Coho Salmon (14.5 in.)**	Weeden's Creek	0.9
4/28/78	6 Creek Chubs (6 in.)**	Weeden's Creek	150.0
4/28/78	18 Minnows (5.0 in.)**	Weeden's Creek	61.0
Kohler Dam Samples			
4/26/78	1 Carp (25 in.)*	Above Kohler Dam-Kohler	240.0
4/26/78	1 Carp (25 in.)*	Above Kohler Dam-Kohler	180.0
4/26/78	1 Carp (22 in.)*	Above Kohler Dam-Kohler	150.0
4/26/78	1 Carp (25 in.)*	Above Kohler Dam-Kohler	250.0
4/26/78	3 Carp (22-25 in.)**	Above Kohler Dam-Kohler	350.0
4/26/78	1 Carp (27 in.)*	Above Kohler Dam-Kohler	250.0
4/26/78	5 Carp (18.5-20 in.)**	Above Kohler Dam-Kohler	460.0
4/26/78	3 Carp (21.5-22 in.)**	Above Kohler Dam-Kohler	320.0
4/26/78	5 Suckers (12.5-13 in.)*	Above Kohler Dam-Kohler	88.0
4/26/78	5 Suckers (9-10.5 in.)**	Above Kohler Dam-Kohler	130.0
4/26/78	5 Suckers (10.5-11 in.)**	Above Kohler Dam-Kohler	39.0
4/26/78	5 Suckers (10-11 in.)*	Above Kohler Dam-Kohler	40.0
4/26/78	5 Rock Bass (4.5-7.5 in.)**	Above Kohler Dam-Kohler	190.0
4/26/78	12 Common Shiners (5 in.)**	Above Kohler Dam-Kohler	100.0

(continued)

*Indicates boneless fillet sample
**Indicates whole fish sample
***ppm - parts per million

TABLE 4. (Continued)

Sampling Date	Species and Average Length	Sampling Location	PCB Concentration µg/g (ppm)***
Sheboygan Lagoon Samples (above Sheboygan Falls)			
4/27/78	38 Bluegill (3.5 in.)**	Sheboygan Lagoon	5.0
4/27/78	10 Redhorse (8.0 in.)**	Sheboygan Lagoon	.5
4/27/78	2 Crappies (6.5 in.)**	Sheboygan Lagoon	2.0
4/27/78	6 Rock Bass (8.0 in.)**	Sheboygan Lagoon	2.0
4/27/78	4 Northern Pike (16.0 in.)*	Sheboygan Lagoon	1.0
4/27/78	7 Redhorse (15.0 in.)*	Sheboygan Lagoon	1.0
4/27/78	1 Carp (25.5 in.)*	Sheboygan Lagoon	1.0
4/27/78	1 Carp (23.5 in.)*	Sheboygan Lagoon	---
4/27/78	4 Carp (17.0 in.)*	Sheboygan Lagoon	1.0
4/27/78	5 Carp (17.0 in.)**	Sheboygan Lagoon	3.0
4/27/78	5 Carp (16.0 in.)**	Sheboygan Lagoon	4.0
4/27/78	8 Carp (15.0 in.)**	Sheboygan Lagoon	2.0
4/27/78	8 Carp (14.0 in.)**	Sheboygan Lagoon	2.0
Johnsonville Dam Samples			
4/28/78	4 Carp (20 in.)**	Above Johsonville Dam	.8
4/28/78	4 Carp (17 in.)**	Above Johnsonville Dam	.8
4/28/78	1 Carp (21.5 in.)*	Above Johnsonville Dam	.4
4/28/78	1 Carp (23.5 in.)*	Above Johnsonville Dam	.6
4/28/78	14 Suckers (10.0 in.)**	Above Johnsonville Dam	.2
4/28/78	9 Rock Bass	Above Johnsonville Dam	.5
4/28/78	3 Crappies (6 in.)**	Above Johnsonville Dam	.7
4/28/78	1 Large Mouth Bass (9.5 in.)**	Above Johnsonville Dam	.4
4/28/78	14 Stonecats (5.0 in.)**	Above Johnsonville Dam	.4
4/28/78	60 Minnows (3.0 in.)**	Above Johnsonville Dam	.4
Mullet River Samples			
5/4/78	5 Carp (13.0 in.)**	Mullet River Samples	.2
5/4/78	5 Carp (12.5 in.)**	Mullet River Samples	.2
5/4/78	1 Sucker (10.5 in.)**	Mullet River Samples	.2
5/4/78	10 Suckers (7.5 in.)**	Mullet River Samples	1.0
5/4/78	14 Bullheads (6.5 in.)**	Mullet River Samples	3.0
5/4/78	9 Creek Chubs (5 in.)**	Mullet River Samples	.2
Onion River Samples			
4/28/78	5 Rock Bass (6.5 in.)**	At Highway V	.2
4/28/78	3 Carp (13.0 in.)**	At Highway A	.4
4/28/78	5 Carp (15.0 in.)**	At Highway A	.2
4/28/78	1 Northern Pike (20.5 in.)*	Below Waldo	.2
4/28/78	5 Carp (13.0 in.)**	Below Waldo	.2
4/28/78	5 Carp (18.0 in.)**	Below Waldo	.2
4/28/78	3 Carp (24.0 in)**	Below Waldo	.2

* Indicates boneless fillet sample

**Indicates whole fish sample

***ppm - parts per million

TABLE 5. PCB CONCENTRATIONS IN SHEBOYGAN RIVER WATERSHED BOTTOM SEDIMENTS

Name	Date of Collection	Date of Analysis	Type of Sample (time)	PCB µg/g (ppm)	Water Depth
Sheboygan River Kiwanis Park	5-9-78	6-7-78	Grab (11:00)	9.6	2.5'
Sheboygan River Between 8th & 14th St., Sheboygan	4-27-78	6-7-78	Grab (10:30)	3.6	7-11'
Sheboygan River upstream from USGS Gauging Station	4-27-78	5-10-78	Grab (13:45)	2.4	4'
Kohler Landfill	4-27-78	5-15-78	Grab (13:15)	1.3	NA
Kohler Landfill	5-9-78	6-7-78	Grab (12:30)	<.2	NA
Sheboygan River upstream from Kohler Dam	4-27-78	5-10-78	Grab (14:30)	52	2'
Sheboygan River upstream from Kohler Dam	4-27-78	5-10-78	Grab (14:30)	81	2'
Sheboygan River upstream from Kohler Dam	5-9-78	6-7-78	Grab (11:50)	27	2'
Sheboygan River 100 feet downstream from Tecumseh	5-9-78	6-7-78	Grab (14:00)	190	3"
Sheboygan River 70 yards upstream from Tecumseh (Sheboygan Falls)	5-9-78	6-7-78	Grab (13:00)	2.4	6"
Upstream from Sheboygan Falls Dam (Sheboygan Falls)	4-27-78	6-7-78	Grab (15:15)	<.2	4'
Upstream from Johnsonville Dam	4-27-78	6-7-78	Grab (19:30)	<.2	2'
Mullet River Hwy. PP	4-27-78	6-7-78	Grab (16:30)	3.4	
Mullet River - Hwy. PP	5-9-78	6-7-78	Grab (15:00)	<.2	2'
Mullet River upstream from Plymouth	4-27-78	6-7-78	Composite (17:45)	<.2	3'
Mullet River below Plymouth STP	4-27-78	6-7-78	Grab (17:00)	<.1	2'
Onion River at Hwy. EE	4-27-78	6-7-78	Grab (15:45)	<.1	3'

TABLE 6. PCB CONCENTRATIONS IN MUNICIPAL AND INDUSTRIAL POINT SOURCES, RIVER WATERS, AND HYDRAULIC FLUIDS

Name	Collection Date	Date of Analysis	Type of Sample (time)	PCB µg/l (ppm)	Comments
Municipal Sewage Treatment Plants					
Sheboygan STP*	3-31-78	4-14-78		<.4	Final Effluent
Sheboygan Falls STP	3-31-78	4-12-78		<1.0	Final Effluent
Kohler STP	3-31-78	4-14-78		<.2	Final Effluent
Plymouth STP	4-4-78	4-11-78		9.0	Final Effluent (Flood Stage)
Plymouth STP	4-27-78	5-15-78		<.2	Final Effluent (Normal)
Belgium STP	4-21-78	5-8-78		<2.0	Final Effluent
Kiel STP	4-24-78	5-8-78		<1.0	Final Effluent
Sheboygan Incinerator	4-21-78	5-8-78		<1.0	Final Effluent
Industrial Effluents					
J.L. French Co. (S)**	4-4-78	4-14-78	Grab	21	Effluent to STP
Thomas Industries (S)	4-4-78	4-7-78	Grab	140	Effluent to STP
Thomas Industries	5-3-78	5-23-78	Grab (11:00)	.2	Surface Water Discharge
Tecumseh Prod. (SF)***	4-4-78	4-20-78	Grab	2.2	Effluent to STP
Tecumseh Prod. (SF)	4-4-78	4-11-78	Grab	<.4	Effluent to Surface Waters
Kohler Co.	4-6-78	4-13-78	Grab	<.2	Effluent to Surface Waters
Kohler Landfill	4-21-78	5-8-78	Grab (09:30)	<1.0	Effluent to Surface Waters
Gilson Co. (P)****	4-21-78	5-8-78	Grab	<.5	Effluent to STP
Ametek (S)	4-24-78	5-8-78	Grab (11:00)	<2.0	Effluent to Surface Waters
Bemis Mfg. (SF)	4-24-78	5-8-78	Grab (14:00)	<.3	Effluent to Surface Waters

(Continued)

*STP - Sewage Treatment Plant
**Sheboygan
***Sheboygan Falls
****Plymouth

TABLE 6. (Continued)

Name	Collection Date	Date of Analysis	Type of Sample (time)	PCB µg/l (ppm)	Comments
Schrier Malting (S)**	4-24-78	5-8-78	Grab (10:00)	<1.0	Effluent to Surface Waters
Vollrath Inc. (S)	4-24-78	5-8-78	Grab (11:30)	<.3	Effluent to Surface Waters
Plastic Engineering (S)	4-24-78	5-8-78	Grab (13:00)	<.3	Effluent to Surface Waters
Plastic Engineering (15th Avenue) (S)	4-24-78	5-8-78	Grab (13:30)	<1.0	Effluent to Lake Michigan
River Water Samples					
Junction of Hwy. 28 & Sheboygan River	4-4-78	4-12-78	Grab	3.0	Flood Stage
Sheboygan River above Sheboygan Falls Dam	4-4-78	4-6-78	Grab	<.5	Flood Stage
Sheboygan River above Sheboygan Falls Dam	4-27-78	5-15-78	Grab (15:15)	<.3	
Sheboygan River above Kohler Dam	4-27-78	5-15-78	Grab (13:25)	<.2	
Sheboygan River below Johnsonville Dam	4-27-78	5-15-78	Grab (19:30)	<.2	
Onion River at Hwy. EE	4-27-78	5-15-78	Grab (15:46)	<.2	
Kohler Landfill	4-27-78	5-15-78	Grab (13:10)	<.3	Leachate
Mullet River above Plymouth Dam	4-27-78	5-15-78	Grab (18:00)	<.2	
Sheboygan River 1/4 mile upstream from USGS Gauging Station	4-27-78	5-15-78	Grab (13:30)	.3	
Sheboygan River & 14th Street (S)	4-27-78	5-15-78	Grab (21:00)	.3	

(Continued)

*STP - Sewage Treatment Plant
**Sheboygan
***Sheboygan Falls
****Plymouth

TABLE 6. (Continued)

Name	Collection Date	Date of Analysis	Type of Sample (time)	PCB µg/g (ppm)	Comments
Sheboygan River immediately downstream from Tecumseh Products	5-10-78	6-7-78	Grab (01:15)	<.4	
Sheboygan River immediately upstream from Tecumseh Products	5-10-78	6-16-78	Grab (01:15)	<.2	
Hydraulic Fluids					
J. L. French Co. (S)**	8-26-78	9-16-78	Grabs	10, 42, 104, 175, 21, 94, 72, 19, 12, 19, 14, 10	Hydraulic Fluids were obtained from on-line aluminum die-cast machines
Thomas Industries (S)	4-4-78	4-19-78	Grabs	.8, 3.3, 3.0	"
Tecumseh Products (SF)***	4-4-78	4-19-78	Grabs	6.0, 2.5, 7.0	"

*STP - Sewage Treatment Plant
**Sheboygan
***Sheboygan Falls
****Plymouth

High PCB concentrations were found in soil samples collected on the Tecumseh property in Sheboygan Falls (Table 7). Granular oil absorbent material deposited on the property contained up to 120,000 µg/g (ppm) PCB (12%). This PBC-contaminated waste desposited on the dike bordering the Sheboygan River is a significant source of PCBs to the lower Sheboygan River. During periods of high water the Sheboygan River flows along the dike and is in direct contact with highly contaminated fill materials. During rains PCBs in the dike are subject to washing into the stream. PCBs at other locations on the plant property are likely to soak into the ground and reach the river through groundwater discharge. The area is subject to periodic flooding which occurred as recently as May, 1978.

Following the discovery of highly contaminated PCB waste deposits at the Tecumseh Products site, the Department of Natural Resources issued an order on May 12, 1978 banning further disposal of solid waste on Tecumseh property. On June 21, 1978 the Department issued a second order requiring the excavation, collection and proper storage of all materials likely to contain PCBs from the dike on the Sheboygan River behind the Tecumseh Plant. PCB-containing materials include oil-absorbent material, scrap pressure hose, and oil-soaked debris.

On June 21, 1978 the Department together with the Department of Health and Social Services lifted the warning against fish consumption in the following sections of the river basin: the Sheboygan River from the Sheboygan Marsh to the Sheboygan Falls Dam, the Mullet River from Plymouth to the junction with the Sheboygan River at Sheboygan Falls and the Onion River from Waldo to Gibbsville.

The PCB warning is currently in effect on the Onion River from Gibbsville to Sheboygan Falls, on the Sheboygan River from the Sheboygan Falls Dam to the Coast Guard Station at Lake Michigan and on two tributaries of the Sheboygan, Weeden's Creek and Greendale Creek. The Onion River, Weeden's Creek and Greendale Creek were included because they are accessible to migrating main channel fish.

At this time Tecumseh Products Company is proceeding with disposal operations. Contaminated wastes from the dike are being placed in sealed drums and stored according to State and Federal regulations until a suitable disposal site becomes available.

INDIANA HARBOR

Indiana Harbor is located at the southern end of Lake Michigan in the State of Indiana near the Indiana - Illinois border. The harbor consists of a breakwater in Lake Michigan 1120 feet long, an entrance channel, an anchorage and maneuver basin, and a ship canal joining the harbor with the Lake George Branch and the Calumet River Branch of the Grand Calumet River (Figure 2).

During 1977 the Chicago District of the U.S. Army Corps of Engineers proposed continued maintenance dredging of Indiana Harbor and respective navigation channels over a five- to six-year period. It was proposed that

TABLE 7. TECUMSEH PRODUCTS SOIL SAMPLES*

Location	Sample Description	Sample Depth	Collection Date Collector	PCB µg/g (ppm)
S-18	8' from east fence, dark oily spot on ground. Pic. 1	topsoil - surface to 2" deep	5-11-78 Sheffy	390.0
S-19	5' from south fence, granular oil absorbent material, alum. bits. Pic. 2	6" deep	5-11-78 Sheffy	120,000.0
S-20	10' from bldg., sandy soil oil dump area oily soil Pic. 3	surface to 3" deep	5-11-78 Sheffy	2,300.0
S-21	10' from bldg., sandy gravelly soil, oil dump area oily sample. Pic. 4	surface to 3" deep	5-11-78 Sheffy	660.0
S-22	3' from bldg., between oil sep. & bldg., black tarry soil sample, oil dump area. Pic. 5	surface to 3" deep	5-11-78 Sheffy	880.0
S-23	3' from south fence, low area below rubble pile, topsoil sample. Pic. 6	topsoil - surface to 4" deep	5-11-78 Sheffy	1,500.0
S-24	Granular oil absorbent material, alum. bits, outside of south fence on dike. Pic. 7	4-8" deep	5-11-78 Sheffy	120,000.0
S-25	Granular oil absorbent material, alum. bits, 2' outside of south fence. Pic. 8	4-8" deep	5-11-78 Sheffy	78,500.0
S-26	Granular oil absorbent material & alum. bits on river side of dike. Pic. 9	4-8" deep	5-11-78 Sheffy	54,000.0
S-27	Granular oil absorbent material & alum bits, hydraulic hose on river side of dike. Pic 10	4-8" deep	5-11-78 Sheffy	43,800.0

*Laboratory analysis for these samples was performed by soaking .5-20 grams of sample in acetone for 1-8 hours. The extract was then injected directly into the gas chromatograph. This procedure is a more rapid procedure for detecting high levels of PCB than the standard column elution method.

750,000 cubic yards of maintenance dredging from the Harbor will be placed in the Inland Steel disposal area during the next six years (3). Proposed dredging will be accomplished using a clamshell. The excavated materials will be placed in barges and towed to a rehandling area located outside of the inland disposal facility. The excavated materials will be mechanically removed from the barges using a clamshell on top of the containment wall and placed in a barge located within the disposal facility. The dredged materials will then be bottom-dumped in the northeast portion of the disposal facility as designated by the Steel Company. A temporary opening will be made in the existing slag dike to provide an access for the towboat and barge(s) which will operate within the disposal facility. The opening will be closed immediately after the equipment has entered the disposal area. The dike will be reopened to permit the equipment to leave the disposal area once dredging operations have been completed. This process will be repeated for subsequent dredging operations.

In 1977 at the request of the Corps of Engineers, the U.S. Environmental Protection Agency evaluated sediment samples in the areas proposed for maintenance dredging of Indiana Harbor.

Sediment samples were collected by the Surveillance and Research Staff of the Great Lakes National Program Office, U.S. Environmental Protection Agency at 13 locations (Figure 2). Analyses were performed by the Central Regional Laboratory of the USEPA, Region V. Macroinvertebrate identification and enumeration was performed by Hiedelberg College, Tiffin, Ohio (4).

Sediments obtained from the upstream limits of the harbor (Stations 1 and 2) to the easterly breakwater (Station 10) were black or dark brown silt containing visible oil and petroleum odor (Table 8). Sediments lakeward of Station 10 were characterized as brown or grey sand and gravel. Macroinvertebrates were identified at these locations during field sampling. Sieve analysis data confirmed sediments which were predominantly silt and clay size at most locations (Table 9).

Bulk sediment analysis was performed on all samples (Table 10). Stations 1 through 11 demonstrated severe contaminant levels with respect to all parameters. The lakeward locations (Stations 12 and 13) showed measurable reductions in mercury, lead, zinc, manganese, and arsenic when compared with measurements taken at other stations.

The elutriate tests show releases of iron, manganese, and aluminum from virtually all samples (Table 11). All samples except those obtained from Stations 12 and 13 showed release of TKN, ammonia, cyanide, and phenols. Arsenic was released from samples at locations 3, 5 (replicate), and 11. Mercury was released from samples at Stations 1 through 5, 8, 12 and 13. In general, samples from locations 1 through 3 exhibited the most release and samples from locations 12 and 13 showed the least release.

Macroinvertebrates were identified at all stations (Table 12). Stations 2, 3, and 4 showed virtual absence of macroinvertebrate taxa. High numbers of the tolerant Oligochaete, Tubifex, were observed at Station 8. Highest diver-

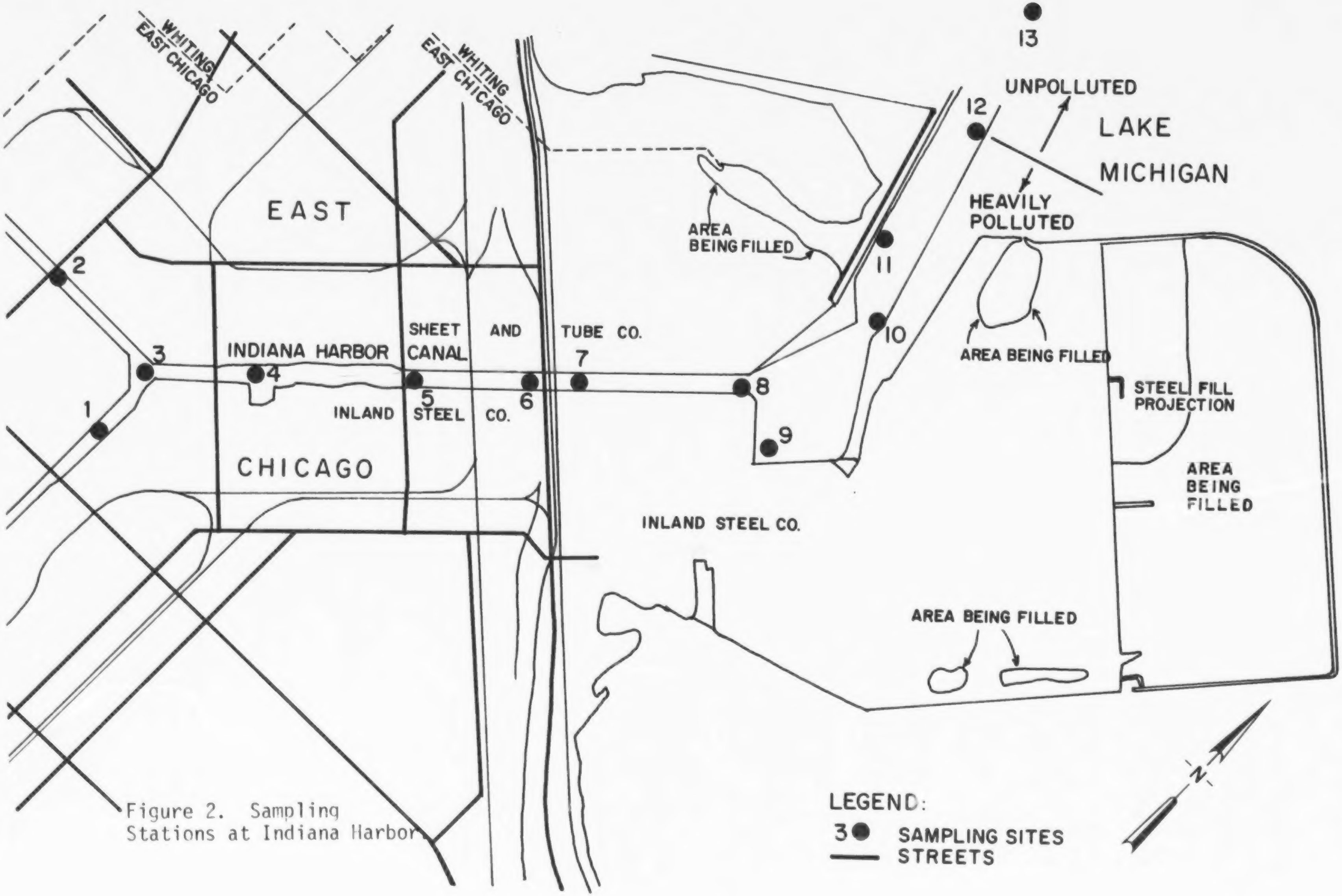
13
12
UNPOLLUTED
LAKE
MICHIGAN
HEAVILY
POLLUTED
WHITING
EAST CHICAGO
WHITING
EAST CHICAGO
EAST
AREA
BEING FILLED
11
10
2
SHEET
AND
TUBE CO.
3
INDIANA HARBOR
CANAL
4
5
6
7
8
AREA BEING FILLED
STEEL FILL
PROJECTION
1
INLAND
STEEL
CO.
9
CHICAGO
AREA
BEING
FILLED
INLAND STEEL CO.
AREA BEING FILLED
N
LEGEND:
3● SAMPLING SITES
STREETS

Figure 2. Sampling Stations at Indiana Harbor

TABLE 8. FIELD OBSERVATIONS AT INDIANA HARBOR

HARBOR: Indiana Harbor, Indiana
SAMPLED: August 30, 1977

Station No.	Depth (ft)	Observations				General Remarks
		Color	Sample Description	Odor	Oil	
1	13	Black	Silt, very oily	Petroleum	Yes	1 grab biology sample
2	14	Black	Silt, very oily	Petroleum	Yes	1 grab biology sample
3	27	Very dark brown	Oily silt	Petroleum	Yes	1 grab biology sample
4	22	Dark brown	Oily silt	Petroleum	Yes	1 grab biology sample
5	22	Dark brown and black	Oily silt, some gravel	Petroleum	Yes	1 grab biology sample
5 (Replicate)						
6	19	Dark brown and black	Oily silt	Petroleum	Yes	1 grab biology sample
7	24	Black	Oily silt	Petroleum	Yes	1 grab biology sample, 1 leech
8	25	Dark brown and black	Silty sand, oily	Petroleum	Yes	1 grab biology sample
9	20	Dark brown and black	Silty sand, oily	Petroleum	Yes	1 grab biology sample
10	22	Dark brown and black	Silty sand, oily	Petroleum	Yes	1 grab biology sample clamshells, sludgeworms
11	26	Dark brown and black	Oily silt	Petroleum	Yes	1 grab biology sample sludgeworms
12	28	Brown and grey	Brown coarse sand, stones, gravel, stiff grey clay	None	None	3 grab biology sample, clamshells, sludgeworms, 1 leech
13	30	Grey	Fine sand	Slight Petroleum	Trace	3 grab biology sample, fingernail clamshells

TABLE 9. SIEVE ANALYSIS AT INDIANA HARBOR

HARBOR: Indiana Harbor, Indiana
SAMPLED: August 30, 1977

Sieve No. and Description	Sediment Size Analysis by Percent at Each Station													
	1	2	3	4	5	5 replicate	6	7	8	9	10	11	12	13
Retained on #10 Medium Gravel and Larger	<1	<1	2	3	62	6	<1	2	4	<1	<1	<1	5	6
Retained on #20 Fine Gravel	<1	<1	<1	<1	10	3	3	<1	5	<1	<1	<1	4	3
Retained on #60 Medium and Coarse Sand	2	4	5	6	13	23	21	2	30	<1	9	5	75	21
Retained on #200 Fine Sand	2	7	4	4	6	20	16	3	10	3	8	4	11	28
Passing #200 Silts and Clays	96	89	89	87	9	48	60	93	51	97	83	91	5	42

TABLE 10. BULK SEDIMENT ANALYSIS AT INDIANA HARBOR

HARBOR: Indiana Harbon, Indiana
SAMPLED: August 30, 1977

Parameter	Station													
	1	2	3	4	5	5 replicate	6	7	8	9	10	11	12	13
Total Solids %	22.8	33.6	41.1	41.5	71.1	63.4	56.5	46.5	61.1	35.1	44.4	47.1	72.0	70.7
Volatile Solids %	24.1	20.3	20.4	10.7	4.75	8.92	12.3	14.8	7.70	13.5	10.4	9.21	1.08	1.38
Chem. Oxy. Demand	520,000	390,000	340,000	190,000	40,000	190,000	170,000	180,000	96,000	220,000	130,000	120,000	8,200	8,200
T. Kjel, Nitrogen	9,500	4,000	3,900	3,100	520	980	1,600	3,100	1,300	2,300	1,700	2,200	140	330
Oil-Grease	80,000	140,000	89,000	28,000	15,000	44,000	35,000	36,000	23,000	76,000	38,000	19,000	<600	<600
Mercury	0.9	1.2	1.2	0.7	0.2	1.4	0.2	0.3	0.3	0.3	0.3	0.1	<0.1	<0.1
Lead	1,300	1,400	1,700	620	370	580	390	560	230	540	410	240	25	33
Zinc	4,200	4,600	10,000	3,000	1,800	4,000	1,500	2,700	860	2,400	2,000	880	150	160
T. Phosphorus	8,700	4,300	6,400	3,600	570	1,500	1,700	2,700	1,400	1,500	1,100	1,100	140	290
Ammonia Nitrogen	1,100	350	660	470	58	85	170	330	98	280	53	120	<10	<10
Manganese	2,100	1,500	3,900	1,400	2,000	1,800	1,800	1,900	6,600	2,000	1,300	1,800	320	320
Nickel	240	100	90	80	65	64	110	80	48	100	47	43	17	7
Arsenic	48	35	41	19	15	39	38	30	21	58	29	18	9	4
Cadmium	22	7	18	4	6	18	17	13	<1	30	6	1	1	1
Chromium	690	1,200	1,700	720	240	410	280	400	190	290	150	160	14	26
Magnesium	7,200	16,000	5,300	20,000	19,000	15,000	12,000	13,000	16,000	8,4000	20,000	24,000	8,700	31,000
Copper	380	290	290	240	150	230	300	270	110	260	120	99	<1	17
Iron	120,000	84,000	210,000	77,000	72,000	120,000	170,000	150,000	110,000	220,000	100,000	93,000	13,000	14,000

All values mg/kg dry weight unless otherwise noted.

TABLE 11. ELUTRIATE TEST RESTULTS FROM INDIANA HARBOR

HARBOR: Indiana Harbor, Indiana
SAMPLED: August 30, 1977

Parameter	Units	Run Blank	Elutriate Water Without Sediment	Elutriate Water Using Sediments at Each Station 1	2	3	4	5	5 replicate	6
COD	(mg/l)	<3	14	31	18	24	16	18	20	26
TOC	"	4	9	16	10	20	11	15	13	12
TKN	"	0.04	3.52	22.1	12.2	15.4	10.9	6.92	5.14	11.4
Ammonia-N	"	0.03	3.02	22	9.6	14.3	9.72	5.23	3.90	9.02
Nitrate+Nitrite-N	"	<0.03	0.22	3.7	10.7	8.9	0.31	0.37	0.23	0.40
Total P	"	<0.02	0.12	0.14	<0.02	0.03	<0.02	0.02	0.02	0.04
Cyanide	"	0.006	0.010	0.021	0.017	0.020	0.022	0.017	0.017	0.020
Phenols	(μg/l)	<5	7	35	25	23	10	10	20	28
Arsenic	"	<2	4	7	4	8	3	6	8	5
Cadmium	"	<20	<20	<20	<20	<20	<20	<20	<20	<20
Chromium	"	<24	<24	29	27	<24	27	41	<24	<24
Copper	"	<4	<4	5	7	5	5	8	<4	<4
Iron	"	496	459	1,780	1,710	1,380	1,580	1,770	1,410	1,420
Lead	"	<60	<60	<60	<60	<60	<60	<60	<60	<60
Magnesium	(mg/l)	1.3	13.2	16.5	18.4	16.8	17.8	18.4	21.1	16.1
Manganese	(μg/l)	<10	35	127	220	134	105	158	116	119
Mercury	"	<0.1	<0.1	0.3	0.6	0.8	2.8	0.1	0.8	0.1
Nickel	"	<50	<50	<50	<50	60	53	111	<50	<50
Zinc	"	<25	32	76	538	115	561	105	92	73
Aluminum	"	531	518	1,810	2,010	1,700	2,020	1,890	1,340	1,300

(continued)

TABLE 11. Continued

HARBOR: Indiana Harbor, Indiana
SAMPLED: August 30, 1977

Parameter	Unite	Run Blank	Elutriate Water Without Sediment	Elutriate Water Using Sediments at Each Station 7	8	9	10	11	12	13
COD	(mg/l)	<3	14	34	31	21	16	25	14	18
TOC	"	4	9	19	15	14	10	11	8	8
TKN	"	0.04	3.52	14.3	8.74	10.3	4.15	6.80	2.98	3.24
Ammonia-N	"	0.03	3.02	11.9	7.56	8.88	3.44	6.15	2.32	2.63
Nitrate+Nitrite-N	"	<0.03	0.22	0.29	0.31	0.33	0.27	0.22	0.32	0.27
Total P	"	<0.02	0.12	0.04	0.04	0.02	<0.02	0.08	0.03	<0.02
Cyanide	"	0.006	0.010	0.017	0.022	0.017	0.025	0.029	0.019	0.016
Phenols	(μg/l)	<5	7	25	23	27	16	14	9	8
Arsenic	"	<2	4	7	5	6	5	10	3	4
Cadmium	"	<20	<20	<20	<20	<20	<20	<20	<20	<20
Chromium	"	<24	<24	<24	<24	<24	<24	<24	<24	<24
Copper	"	<4	<4	<4	<4	<4	<4	<4	<4	<4
Iron	"	496	459	1,250	1,090	1,680	1,420	1,900	1,730	1,660
Lead	"	<60	<60	<60	<60	<60	<60	<60	<60	<60
Magnesium	(mg/l)	1.3	13.2	17.1	17.5	19.2	15.9	19.5	17.0	15.5
Manganese	(μg/l)	<10	35	121	288	136	106	334	39	228
Mercury	"	<0.1	<0.1	0.1	0.4	0.1	0.1	0.1	0.2	0.2
Nickel	"	<50	<50	<50	<50	<50	<50	<50	<50	<50
Zinc	"	<25	32	43	64	54	49	62	35	41
Aluminum	"	531	518	999	879	1,730	1,340	2,020	1,940	1,750

TABLE 12. MACROINVERTEBRATE DATA AT INDIANA HARBOR

HARBOR: Indiana Harbor, Indiana
SAMPLED: August 30, 1977

	Number of Organisms for Each Taxa by Station												
Taxa	1	2	3	4	5	6	7	8	9	10	11	12	13
Diptera													
Chironomus	9												2
Micropsectra	1												
Microtendipes	1												
Dicrotendipes	2					1			1				
Kiefferulus	2												
Procladius	3											1	
Brillia	1												
Chaoborus		1											
Chryptochironomus												1	2
Trichocladius													3
Oligochaeta													
Tubifex	77				210	59	76	1,400	27	43	176	375	31
Limnodrilus	44				281	20	95	290	5	4	25		
Peloscolex	3				23		43	650	9	20	51		3
Branchiura sowerbyi								10					
Unidentifiable immature					45	8	24	150		30	35	25	8
Hirudinea													
Glossiphonia							1						3
Macrobdella													
Haemopis											1		
Helobdella													1
													4
Amphiopoda													
Gammarus													1
Pelecyopoda													
Pisidium				1									
Sphaerium				3		6	17	41	7	35	6		85
Muscueium								17			3		15
Gastropoda													
Lymnaea			1	2					2				2
Physa			1	4		21	4	16	48		8		4
Goniobasis				1	1			2					
Helisoma										3	1		
Gyraulus													7
Total # of organisms	143	1	2	11	560	114	260	2576	99	135	306	402	171
Total # of taxa	10	1	2	5	5	5	7	9	7	6	9	4	15

sity of organisms was measured at Station 13. High diversity and higher numbers of Pelecepods and Gastropods may be attributed to suitable fine sand substrate and lower bottom sediment contaminant levels at this location.

The bulk sediment chemistry, PCB (polychlorinated biphenyls), and pesticides results show all compounds except PCBs were below the laboratory's detection limits (Table 13). The detection limits for some compounds were higher than usual due to high interferences present in the samples. Low PCB concentrations were detected at Stations 12 and 13. Elevated levels of PCBs were found at Station 1, and Stations 6 through 11. High concentrations of PCBs ranging from 17.9 mg/kg to 25.7 mg/kg were detected at Stations 2 through 5.

CONCLUSION AND RECOMMENDATIONS

In April 1978, the Region V office of the U.S. Environmental Protection Agency issued the following position on maintenance dredging of Indiana Harbor (3):

> Analysis of Indiana Harbor indicated that bulk sediment concentrations of toxic constituents such as polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), mercury, lead, arsenic, cadmium, and chromium and nutrient concentrations are all high. Elutriate testing shows large releases of total Kjeldahl nitrogen (TKN), ammonia, iron, manganese, nitrate and nitrite, phenols, zinc, and aluminum from virtually all samples. Some releases of cyanide, arsenic, chromium, copper, mercury, and nickel are also indicated.
>
> There are three municipal drinking water intakes within three miles of the Indiana Harbor mouth. PCB levels in Lake Michigan are already at problem levels, and Lake Michigan is experiencing eutrophication problems, particularly in the southern end. Utmost care will have to be taken to ensure against adverse environmental health effects from release of toxic and carcinogenic substances due to dredging and subsequent disposal of sediments from Indiana Harbor. Consequently, the following contingencies must be incorporated into proposed plans for dredging and disposal of bottom sediments for USEPA approval of the proposed action.

DREDGING AND DISPOSAL REQUIREMENTS

1. The dredging must not result in any increase over background water concentrations (on whole-water, unfiltered sample basis) as measured between the inner U.S. lights (when dredging upstream of that point) for mercury, lead, arsenic, cadmium, chromium, cyanide, PCB, PAH, TKN, ammonia, nitrate and nitrite, total phosphorus, BOD (biochemical oxygen demand), phenols, copper, zinc, aluminum, chloride, and oil and grease. When dredging downstream of the inner U.S. lights, these same limitations should be applied as measured between the ends of the outer breakwaters.

2. During dredging, no visible debris or oil slick should extend beyond the ends of the outer breakwaters at any time.

TABLE 13. BULK SEDIMENT CHEMISTRY PCB AND PESTICIDES ANALYSIS (All values are mg/kg/dry weight).

HARBOR: Indiana Harbor, Indiana
SAMPLED: August 30, 1977

	Sample Site													
Compound	1	2	3	4	5	5 replicate	6	7	8	9	10	11	12	13
Hexachlorobenzene	*	*	*	*	*	*	*	*	*	*	*	*	*	*
beta Benzenehexachloride	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Lindane	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Treflan	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Aldrin	<0.3	<0.9	<0.7	<1.0	*	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	*	*	*
Isodrin	<0.2	<1.3	<0.5	<1.2	*	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	*	*	*
Heptachlor Epoxide	<0.15	<1.0	<0.35	<0.7	*	*	*	*	*	*	*	*	*	*
gamma Chlordane	<0.2	<0.2	<0.2	<0.2	*	*	*	*	*	*	*	*	*	*
o,p -DDE	*	*	*	*	*	*	*	*	*	*	*	*	*	*
p,p'-DDE	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	*	*	*	*	*	*	*	*
o,p -DDD	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
o,p -DDT	*	<0.5	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	*	*	*	*	*
p,p'-DDD	*	<0.2	*	*	*	*	*	*	*	*	*	*	*	*
p,p'-DDT	*	<0.5	*	*	<0.65	*	<0.3	<0.3	<0.3	<0.3	*	*	*	*
Methoxychlor	<0.2	<2.0	<0.2	<0.2	<1.7	*	*	<1.4	*	*	*	*	*	*
Mirex	*	<0.2	*	*	*	*	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
2,4-D, Isopropl Ester	<0.5	<0.5	<0.5	<0.5	<0.5	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Endosulfan I	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Dieldrin	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
Endrin	*	*	*	*	*	*	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
Endosulfan II	<0.02	<0.02	<0.02	<0.02	<0.02	*	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
DCPA	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03
Tetradifon	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Aroclor 1016 (1242)	5.6	21.3	25.7	20.9	23.1	17.9	6.5	9.5	3.4	4.4	1.5	1.4	0.043	0.042
Aroclor 1248	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Aroclor 1254	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.56
Aroclor 1260	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Total PCB	5.6	21.3	25.7	20.9	23.1	17.9	6.5	9.5	3.4	4.4	1.5	1.4	0.043	0.098

*Concentration less than 0.1 mg/kg.

3. The dredging should not result in any increase over background water concentrations for whole-water, unfiltered samples taken in the raw water of the three drinking water intakes for the parameters cited in item 1 above.

4. The U.S. Army Corps of Engineers (COE) should consult with appropriate sources to arrive at a proposed dredging method which will meet the dredging requirements. It should be substantiated, through modeling and a test situation, that the proposed dredging method will meet the dredging requirements.

5. In order to determine whether the proposed method of dredging is likely to meet the dredging requirements, the dredging contractor should test dredge at typical levels of production in the area upstream of the inner U.S. lights. Dredging should then be suspended until sampling results are reported and reviewed.

6. Disposal operations must include chemical fixation of sediments. An investigation will have to be conducted to determine the most effective stabilizing agent and optimum dosage for the type of sediments involved and the environmental conditions that will be encountered.

7. Disposal operations must include construction of another containment facility within the existing inland facility. The inner facility would have to be designed to be impermeable, with a weir and whatever controls for treatment may be necessary for effluent discharges. The inner facility should be large enough to retain sediments without discharge until applicable water quality standards can be met. Water quality should be monitored in the area between the two dikes, as well as outside of the existing (outer) facility, during disposal operations. An initial disposal testing period must be established to determine the effectiveness of the inner-dike system.

8. Depending on the type of disposal method selected, a water quality monitoring plan will have to be developed for the disposal area.

The U.S. Army Corps of Engineers and the U.S. Environmental Protection Agency are currently reviewing these recommendations as well as other options for maintenance dredging of Indiana Harbor. An alternative disposal site is also under consideration.

SAGINAW RIVER AND BAY

During 1977, an overview of PCB contamination in the State of Michigan of Saginaw River and Bay was presented at the Third U.S./Japan Experts' Meeting on Management of Bottom Sediments Containing Toxic Substances (6).

The presence of PCBs in the Saginaw River and Bay resulted in contamination of fish populations. The Michigan Department of Agriculture detected PCB levels in channel catfish in excess of the U.S. Food and Drug Administration tolerance level of 5 mg/kg. As a result, a ban was issued for commercial catfishing in the inner Saginaw Bay area. The continued detection of PCBs in Saginaw Bay fish indicated that PCBs previously released to sediments continue to be detected in resident fish populations.

In view of existing PCB problems in Saginaw River and Bay, the U.S. Environmental Protection Agency was unable to concur with proposed maintenance dredging and disposal in 1976 until a complete sediment analyses program for PCBs was undertaken.

The U.S. Environmental Protection Agency (Chicago Office) and the U.S. Army Corps of Engineers (Detroit District) conducted an intensive survey of Saginaw River and Bay in October, 1976. PCBs were sampled at 35 locations in Saginaw River and at 11 locations in Saginaw Bay. PCB concentrations in bottom sediments ranged from <0.1 to 22.9 mg/kg for all locations. Samples taken in the downstream vicinity of the City of Saginaw Wastewater Treatment ranged from 5.5 to 22.9 mg/kg PCB. PCB concentrations near the Bay City Waste Water Treatment Plant ranged from 3.5 to 11.8 mg/kg. Sediment samples obtained in Saginaw Bay had considerably lower PCB concentrations ranging from 1.3 to 4.2 mg/kg.

During 1978, the U.S. Environmental Protection Agency requested that the Michigan Department of Natural Resources review the two PCB-affected areas, downstream of the Saginaw Wastewater Treatment Plant and in the vicinity of the Bay City Wastewater Treatment Plant. The Department of Natural Resources indicated that a metal casting plant in the Saginaw area and an automotive motor facility in Bay City were possible sources. Both facilities had hydraulic systems that were previously contaminated with PCBs. PCB hydraulic fluids were initially flushed at both plants in 1971 and were replaced with PCB-free hydraulic fluids. Samples from the two facilities and the wastewater treatment plants are currently being analyzed.

During 1978, maintenance dredging was conducted at the Saginaw River channel by the Detroit District Corps of Engineers. Dredging was accomplished by the Hopper Dredge HAINS from areas of low level PCB contamination (<10 mg/kg). Dredged materials were disposed of in the Middle Grounds Disposal Site in Bay City, Michigan. It has been proposed that subsequent dredging of more highly contaminated areas will use the recently completed Saginaw Bay Dike Disposal Facility.

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DEVELOPMENTAL ASPECTS AND CURRENT POLICIES FOR RESTORATION AND PROTECTION OF PUBLICLY OWNED FRESHWATER LAKES IN THE UNITED STATES

by

Spencer A. Peterson
Corvallis Environmental Research Laboratory
U.S. Environmental Protection Agency
Corvallis, OR 97330

and

Robert J. Johnson
Criteria and Standards Division
U.S. Environmental Protection Agency
Washington, D.C. 20460

ABSTRACT

Some historical aspects of the impact of pollution on lake water quality in the United States are discussed. The results of two nationwide lake water quality surveys are addressed and the development of the United States Government's involvement in water pollution control is traced from 1912. Although restoration of degraded lakes and protection of higher quality lakes was authorized by the Federal Water Pollution Control Act Amendments of 1972, the "Clean Lakes Program" received relatively low priority for almost seven years. Other, more geographically pervasive pollution control programs were determined more important. In fiscal year 1979, as a result of strong public demand, the Environmental Protection Agency sought funds and positions to support the program. Thus, the formulation of National Lake Restoration policy in the United States has been gradual. A regulation is proposed by the EPA, Office of Water Planning and Standards to facilitate more uniform and equitable administration of the Clean Lakes Program.

INTRODUCTION

The United States has more than 100 thousand lakes ranging in size from a few hectares to Lake Superior at 8.41×10^6 hectares. The State of Minnesota is known as "the land of 10,000 lakes" which is really a misnomer since the state actually has somewhat in excess of 13,000 lakes. In the past, the large number of lakes in the United States has produced a certain amount of apathy among their users. The attitude has been that lakes were there to be used, and Americans have used them with little regard for their sometimes delicate ecosystems. Lakes have served as drinking water supplies and for fishing,

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swimming, boating and various other recreational uses. Unfortunately they have also served as convenient recipients of municipal and industrial wastewater.

Historically, the latter uses primarily impacted urban lakes; however, the problem of water quality degradation of both urban and rural lakes became more pronounced in the United States after 1945 when phosphate-based detergents were introduced. Within a few years the phosphorus content of domestic wastewater doubled and the adverse effect on urban lakes became more severe. About the same time, Americans began to enjoy new freedoms through improved working conditions and reduced working hours. A 40-hour work week became commonplace as did the private ownership of automobiles. Largely due to these two factors, coupled with the introduction of phosphate detergents and larger, more concentrated populations, the problem which had been generally contained to urban lakes began to impinge on an increasing number of the Nation's small rural lakes. By 1972, 92.8% of the vacations and 60.9% of the overnight recreation trips in the United States involved round trips of 400 miles or more (USDA, 1973). Reduced working hours meant increased leisure time. Privately owned cars brought greater mobility and higher living standards permitted second home ownership for many. Lake fronts were among the most popular locations for these second homes. By the mid 1950's cabins and homes completely ringed the shorelines of many previously remote and pristine lakes. Too frequently, the wilderness weekend retreat was simply an extension of everyday urban and surburban living. The weekend recreationist washed his car, clothes, dishes, etc. with high phosphate detergents, flushed his modern toilet and otherwise overloaded inadequate septic wastewater treatment facilities, thus promoting the flow of enriched groundwater to local lakes (Ellis and Childs, 1973; Dudley and Stephenson, 1973). Lakeside lawn fertilization and gardening also produced adverse effects. The situation worsened as lifestyles continued to change and increasing numbers of people made lakefront homes their primary residences. Recreational experiences became a way of life in the United States.

Recreation was not the only cause of small-lake water quality problems. Intensified agricultural practices contributed nutrients, pesticides, herbicides, and silt to the Nation's freshwater streams and lakes. The overabundance of plant nutrients, however, is the most evident of these problems, since the nutrients support massive growths of algae and rooted aquatic plants which hamper the recreational uses of lakes.

EXTENT AND NATURE OF THE PROBLEM

The fact that a declining water quality condition in lakes was associated with increased use of phosphorus-based detergents and intensified farming practices was not coincidental. There is abundant literature to demonstrate highly significant correlations between algal concentrations in lakes and the phosphorus content of phosphorus-limited water. An excellent coverage of the subject is presented in a paper by Nicholls and Dillon (1978).

In 1971 Ketelle and Uttormark (1971) surveyed problem lakes of the United States. Their primary objective was to identify those lakes that had deteriorated so much that rehabilitation would be required if satisfactory water quality was to be re-established. Forty of the 48 contiguous States responded to questionnaires distributed by the University of Wisconsin. Through this process, approximately 400 problem lakes were identified. That number cannot be considered indicative of the problem since some States failed to reply (no lake problems) while others submitted extensive lists (49 problem lakes in Florida). Another problem with the survey was that it left individual respondents with the responsibility for determining which lakes had deteriorated to the extent that restoration would be required.

The survey conducted by Ketelle and Uttormark did point out a number of significant factors. Of the 40 States that responded, 35 indicated they had some lakes with severe water quality problems (Table 1). This indicates the problem was widespread. Although 10 different problem types were listed by various States, the two most frequently cited were nuisance algal blooms and nuisance rooted aquatic plants. The most frequent problem source was municipal effluent (a point source), cited nearly 2 to 1 over the next most common individual problem source. When taken collectively, however, the non-point source problems were cited more frequently than the point source problems.

Another water quality survey was initiated in early 1972 by the Environmental Protection Agency. This survey stemmed largely from the then active controversy over the removal of phosphates from detergents due to their alleged adverse impact on freshwater lakes and reservoirs. The objectives of the National Eutrophication Survey (NES) were to:

(1) Identify those lakes and reservoirs in the contiguous United States that received nutrients from discharges of municipal waste treatment facilities.

(2) Determine the effect of those point source nutrient inputs on the nutrient levels and primary productivity of the water bodies, and

(3) On the basis of the survey information, to advise the Construction Grants Program (building of wastewater treatment facilities) on the cost effective allocation of Federal funds for the construction of tertiary waste treatment facilities for phosphorus removal.

Under these guidelines the NES sampled 401 lakes east of the Mississippi River and not surprisingly found that 78% of them were eutrophic according to the 6 parameter trophic classification system (total phosphorus, chlorophyll a, Secchi depth, dissolved orthophosphorus, inorganic nitrogen and hypolimnetic oxygen saturation percentage) used by NES (USEPA, 1974). West of the Mississippi River the objectives of the survey were broadened to assess nonpoint source (NPS) contributions of nutrients to the lakes and to assist in establishing water quality criteria for nutrients (Allum, Glessner and Gakstatter, 1977). After sampling 374 lakes in the western United States, the National Eutrophication Survey determined that 72% were eutrophic.

Lakes contribute substantially to the economy of states which promote water-oriented recreation. Although accurate cost figures are not readily available it is evident that eutrophication and water pollution impairs the water-oriented economy of these areas due to lower property values, cost of restoring or managing the lakes, and their generally reduced usability. Bartsch (1972) indicated in 1972 that the loss amounted to several million dollars per year. This figure no doubt has increased substantially since that time.

LEGISLATION AND CONTROL PROGRAMS

LEGISLATION

The U.S. Federal Government's involvement in the water pollution control field dates back to 1912 and has its roots in the U.S. Public Health Service. There was limited activity in this area, however, until 1948 when the Water Pollution Control Act of 1948 required the Public Health Service to provide technical information to States. Even then pollution control measures were not of high priority at the Federal level. A series of Congressional Acts up through 1956 established public support for Federal responsibility in this area, but the Acts were generally ineffective. According to Reitze (1972), "The new Acts were designed to be unenforceable (industry doesn't want control, the public does. Compromise½ Pollution control laws for the public, unenforceability for industry.)". The Water Pollution Control Act Amendments of 1961 (P.L. 87-88, 1961) were significant with regard to the Federal Government's involvement in water pollution control since the Amendments authorized the establishment, equipping, and maintaining of field laboratories at 7 locations in the United States. One of those laboratories was located at Corvallis, Oregon.

Finally in 1965, the Water Quality Act (P.L. 89-234, 1965) provided the enabling legislative framework to remove the responsibility for water pollution control activities from the Public Health Service and to establish the Federal Water Pollution Control Administration (FWPCA). Reitze (1972) has pointed out that the Federal Water Pollution Control Administration's chief concern became the dispensing of large amounts of Federal dollars to construct sewage treatment plants. Approximately 90% of their budget went for this purpose; only 10% supported the administration, enforcement programs, research and development, planning, training and technical assistance activities. In spite of the limited research efforts of the Federal Water Pollution Control Administration and largely due to the efforts of Dr. A. F. Bartsch, the National Eutrophication Research Program was established in 1968 at what is now the Corvallis Environmental Research Laboratory (CERL). This laboratory eventually became the focal point for freshwater ecological effects research within the Environmental Protection Agency. Information developed by the Corvallis Research Laboratory and its contractors played a key role in shaping the future of freshwater lake pollution control in the United States.

In 1969 the public's concern for environmental conditions culminated in a demand for action by the U.S. Congress. Several pieces of new legislation and modifications of older legislation began to appear. The Water Quality Improvement Act of 1970 (P.L. 91-224, 1970) allowed conversion of the Federal Water Pollution Control Administration to the Federal Water Quality Administration (FWQA), which, in 1971, became part of the newly created Environmental Protection Agency (EPA). About that same time the National Eutrophication Research Program at CERL was upgraded to the National Eutrophication and Lake

Restoration Program. The first lake restoration research project to be funded by EPA under this program was located at Shagawa Lake, Minnesota (Malueg et al. 1973) where a full-scale advanced wastewater treatment system was built. It was designed to remove phosphorus from domestic wastewater down to a level of 0.05 mg/l. In 1972 Congress passed the most far-reaching piece of legislation to date concerning water pollution control measures, the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500, 1972). The most significant aspect of this law with regard to lakes was Section 314 which states that:

> A. Each State shall prepare or establish, and submit to the Administrator (of the U.S. Environmental Protection Agency) for his approval
>
> 1. An identification and classification according to eutrophic condition of all publicly owned freshwater lakes in such State;
>
> 2. Procedures, processes, and methods (including land use requirements), to control sources of pollution of such lakes; and
>
> 3. Methods and procedures, in conjunction with appropriate Federal agencies, to restore the quality of such lakes.
>
> B. The Administrator shall provide financial assistance to States in order to carry out methods and procedures approved by him under this section.

The law further specified that the amount granted to any State for any fiscal year to conduct approved methods and procedures under this section would not exceed 70% of the total project cost. The Act authorized $50 million for the fiscal year ending June 30, 1973, $100 million for 1974 and $150 million for 1975 for grants to States to conduct work to comply with the various provisions of the law.

A number of other sections of the 1972 amendments to the Act would eventually impinge on lake water quality. One of the most important was Section 208 which required the development of areawide waste treatment management plans. It specified that States would identify areas which, as a result of urban and industrial concentrations or some other factor, had substantial water quality problems. It directed the States to then develop an organization which would put into operation a continuing areawide waste treatment management planning process. This program would be comprehensive in coverage of pollutant types and sources, but it identified specifically that agricultural and silvicultural related non-point sources of pollution be identified and that, to the extent feasible, methods be set forth to control the problem. The approach of dealing with these and other non-point sources of pollution coincides with the most frequently cited problem sources shown in Table 1. Therefore, the Section 208 program will eventually have a significant impact on a number of freshwater lakes, particularly those in urban settings. While

TABLE 1. PROBLEM LAKES OF THE UNITED STATES SHOWING FREQUENCY OF PROBLEM TYPES AND SOURCES OF POLLUTANTS (Modified from Ketelle and Uttormark, 1971)*

Lake Location (State)	Total Surface Area Impacted (ha)	Number of Lakes Impacted	Frequency of Occurrence																	
			Problem										Source							
			Nuisance algal growth	Nuisance aquatic plants	Fish kills	Bacterial contamination	Toxic contamination	Oil brines	Unstable water levels	Siltation	Excessive solids	Unspecified eutrophic conditions	Municipal effluents	Septic tanks	Feed lots	Rural runoff	Urban runoff	Industrial wastes	Ground water	Marsh drainage
Arkansas	1,214	1					1			1				1		1				
California	167,552	4	2							1	2		2	1		1	1			
Colorado	4,431	4				3				1	1		3	3		1		1		
Connecticut	1,801	8	6	5									1							
Delaware	1,158	41	14	33					32				16							
Florida	251,519	49	35	11	18	14	1		1	16	1	5	25	3	1	16	21	16	4	3
Georgia	2,108	2	2	1	1	1							1							
Illinois	2,590	1	1										1	1		1				
Indiana	2,551	29	17	17	2	4			2	1			12	3	1	11	1	2		7
Louisiana	274,082	11				1	2	6		2						1	1	8		
Maine	11,111	21	20	14		2							19			7		6		
Maryland	32	1	1							1			1	1			1			
Massachusetts	2,229	23	9	18	2		1					3	5	6	2	3		2		
Michigan	7,596	24	15	6		1		1		1		3	17	2		2		6		
Minnesota	21,659	29	27	8					6				12	8	1	3	1	2	1	
Missouri	392	1	1											1	1	1				
Nebraska	1,096	3	3	3												3				
New Hampshire	1,813	2											2							
New Jersey	2,562	26	13	20	1		1	1		7										
New York	76,295	10	8	4	2					1		1	7	3				2		
North Dakota	645	2	2	1									1		1	1				
Ohio	10,013	11	1	3				1		7			1	1				3		
Oklahoma	28,634	4	2			1					1	1	1	3	1	1	1	2		
Oregon	41,621	3	3	1		1								2						
Pennsylvania	10,220	14	3	12	1	1				1			3	10		5	3	3		1
South Dakota	18,430	10								3				3	2	3				
Vermont	120,823	3	1		1	1	1					2	2	1				1		
Virginia	1,358	4	2		2							1	3				1	1		
Washington	10,747	19	15	1	1							4	3	4			1	2		
Wisconsin	87,872	34	24	21	8	2			6			5	17	10	1		15	7	8	
Totals	1,164,154	394	227	179	39	32	7	9	47	43	5	25	155	67	11	61	47	64	13	11

* Those states not reporting the number of surface acres impacted or the nature and source of the problems were omitted from the original table by Ketelle and Uttormark.

the Areawide Waste Treatment Management Program will be important in protecting, and to some degree restoring, lakes by limiting the input of pollutants, the Section 314 Clean Lakes Program will be primarily responsible for restoring degraded lake systems to an acceptable level and protecting others from degradation.

CONTROL PROGRAMS

Section 314 of PL 92-500 is known as the "Clean Lakes Section" and the EPA program associated with it has come to be known as the "Clean Lakes Program". Basically, it is a Federal/grantee cost sharing program designed to improve the water quality and general usability of the Nation's freshwater lakes. The program, historically, has received a relatively low priority ranking among what the Environmental Protection Agency has considered more critical programs. Among these are the Construction Grants Program to build waste treatment plants, the Areawide Waste Treatment Management Program, the development of effluent guidelines to control industrial discharges, and more recently the Toxic Substances Control Program.

EPA entered the lake restoration field in FY 1975 when Congress independently appropriated $4 million for expenditure on the Clean Lakes Program. EPA's Office of Water Planning and Standards was charged with dispensing these funds for public use.

POLICIES FOR RESTORATION AND PROTECTION OF FRESHWATER LAKES

EARLY POLICY AND LEGISLATIVE AMENDMENT

Policies for implementing the Clean Lakes Program have evolved gradually. One of the first policy decisions concerned the Federal/grantee cost sharing fractions. Although authorized by Section 314 of the law to Federally fund lake restoration projects up to 70%, the Environmental Protection Agency made an early decision to limit such funding to 50% of the total project cost.

Another early problem was the fact that EPA did not have a funding mechanism to disperse funds appropriated under the authority of Section 314. The Agency did have, however, an established regulation to support research and demonstration grants. Therefore a policy was established to fund lake restoration projects through that mechanism. This decision eventually caused a good deal of confusion and will be addressed later in this paper, however, it was a most reasonable decision at the time considering the state-of-the-art of lake restoration. Lake restoration technologies were predictable qualitatively, but not quantitatively; that is, the direction of water quality change caused by applying a certain technology was known, but not the degree of change.

Another significant decision by the Agency was to make a portion ($400,000) of the initial $4 million lake restoration appropriation available to the Office of Research and Development to assess the overall effectiveness of various lake restoration techniques being used in the Clean Lakes Program. Responsibility for this aspect of the program was transferred to the Corvallis Environmental Research Laboratory where scientists developed an experimental design to accomplish the objectives of the evaluation program. The design was approved by the Office of Water Planning and Standards and implemented in conjunction with the lake restoration program. Descriptions of the experimental design and approaches to the evaluation program have been reported in a number of papers (Porcella and Peterson, 1977; Porcella, Peterson and Glessner, 1977; Peterson, 1978; Porcella, Peterson and Larsen, 1978; and Honey and Hogg, 1978).

The early policy for reviewing grant applications is illustrated in Figure 1. According to this procedure, the applications had to be consistent with State and Federal areawide water quality planning programs. All applications were reviewed by the Office of Water Planning and Standards (OWPS) staff members, at least two university consultants, and by a staff member of the Corvallis Environmental Research Laboratory (representing the Office of Research and Development). The review panel, consisting of members from each of these groups as well as EPA's Regional Office Lake Restoration Coordinators, met when deemed necessary by the Office of Water Planning and Standards. The merits and demerits of each proposal were discussed and the committee made a decision to either fund, reject or request additional information on a given proposal. In late 1975 a number of projects were funded using this review process and the initial $4 million appropriated for the program.

In 1976, the Office of Water Planning and Standards received another $15 million from Congress to meet the growing demand for lake restoration dollars. Again, part of these funds ($1,500,000) were provided to the Office of Research and Development for evaluation projects. In 1977, Congress reaffirmed its commitment to the Clean Lakes Program by amending Section 314 of Public Law 92-500. The Clean Water Act of 1977 (P.L. 95-217, 1977) extended the time frame of the program through Fiscal Year 1980 and authorized $50 million for it in 1977 and an additional $60 million per Fiscal Year for 1978, 1979 and 1980. The amendments under Section 304(j) also required the Administrator of EPA to issue biennially updated "procedures and processes as may be appropriate to restore and enhance the quality of the Nation's publicly owned freshwater lakes" (PL 95-217, 1977). In 1977, Congress appropriated another $15 million for program operation. The Agency now had a $34 million program.

In 1978 the Congress looked at the needs of the Clean Lakes Program and trimmed its funding level to $2.3 million. A major reason for this reduction in funding was that the program had not obligated several million dollars of the previous year's budget. What the Congress did not realize was that all the funds had been committed. As of April 24, 1978, 65 lake restoration projects in 21 states had been funded (Table 2). During August 1978, the first National Conference on Lake Restoration was sponsored by EPA and hosted by the State of Minnesota. The conference attracted over 450 participants from 39 states and Canada. Increasingly it has become apparent that "Clean Lakes" is a broadly based, grassroots supported program that has high potential for dramatic pollution control and water quality improvement of lakes and their tributaries.

During the FY-79 budgeting process the Agency made its first formal request for funds and positions to support the program. EPA asked for and received $15 million and 15 positions. Of greater significance however was

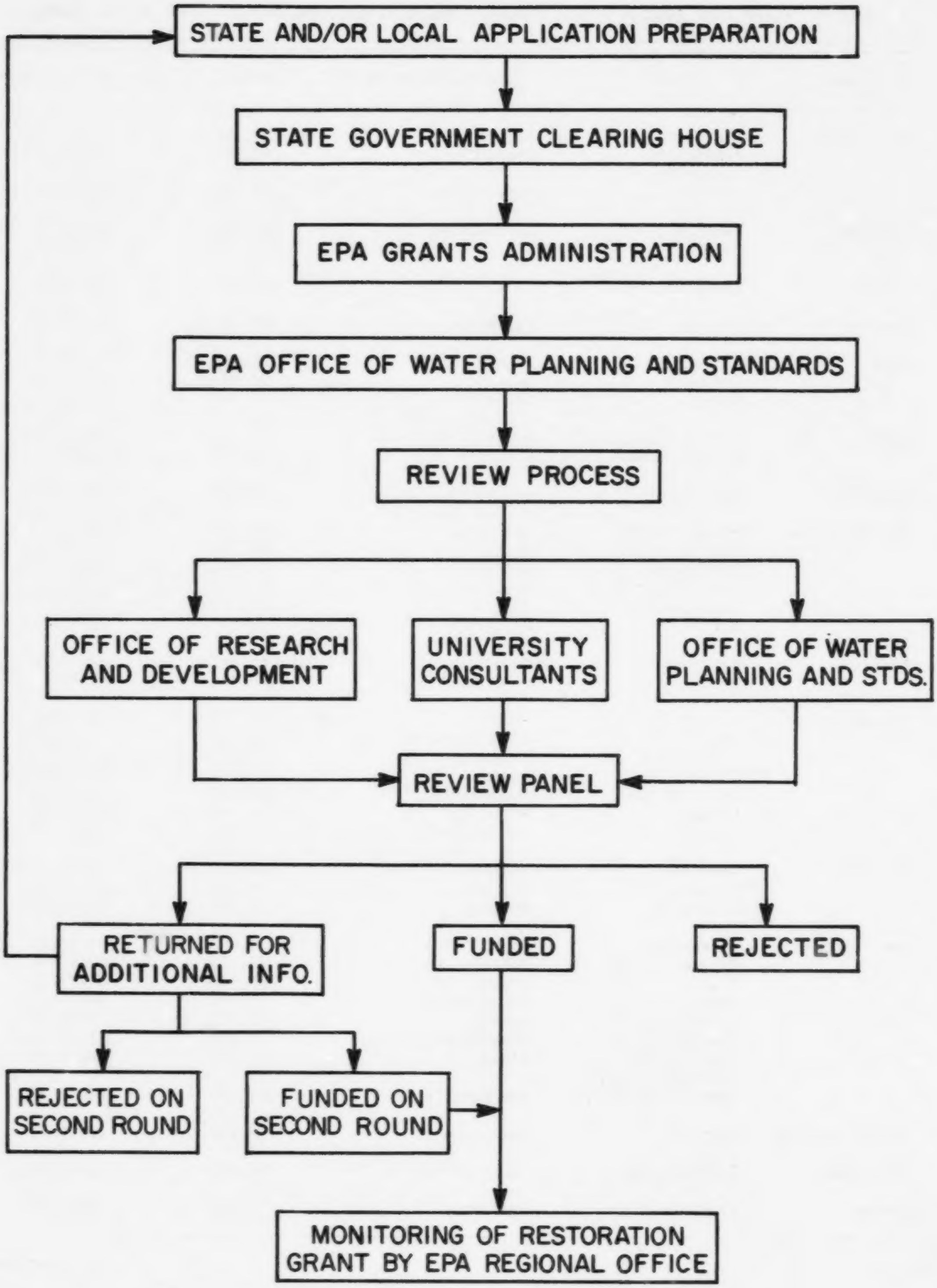
STATE AND/OR LOCAL APPLICATION PREPARATION
STATE GOVERNMENT CLEARING HOUSE
EPA GRANTS ADMINISTRATION
EPA OFFICE OF WATER PLANNING AND STANDARDS
REVIEW PROCESS
OFFICE OF RESEARCH AND DEVELOPMENT
UNIVERSITY CONSULTANTS
OFFICE OF WATER PLANNING AND STDS.
REVIEW PANEL
RETURNED FOR ADDITIONAL INFO.
FUNDED
REJECTED
REJECTED ON SECOND ROUND
FUNDED ON SECOND ROUND
MONITORING OF RESTORATION GRANT BY EPA REGIONAL OFFICE

Figure 1. Clean Lakes application procedures.

TABLE 2. CLEAN LAKE PROJECT SITES, FEDERAL FUNDING LEVELS, AND DATES OF THE INITIAL AWARDS

State	Lake	Grantee/Location	Amount	Date of First Award
California	Stafford*	Novato	$ 290,250	04/08/77
"	Temescal*	Oakland	244,486	03/31/77
"	Lafayette	"	49,250	03/21/77
"	Ellis*	Marysville	1,375,000	05/20/77
Florida	Jackson	Tallahassee	302,834	08/31/76
"	Apopka	"	215,734	01/06/76
Illinois	Frank Holton*	E. St. Louis	927,000	03/16/77
Indiana	Skinner	Ft. Wayne	403,496	09/17/76
Iowa	Lenox*	Lenox	100,000	03/08/77
"	Oelwein*	Oelwein	59,490	03/11/77
"	Blue*	Onawa	372,500	11/10/77
Maine	Little Pond	Damariscotta	11,710	02/19/76
"	Annabessacook	Winthrop	278,020	08/01/77
Maryland	Lock Raven	Baltimore	110,000	04/07/77
Massachusetts	Morses Pond*	Wellesley	308,740	08/23/76
"	Charles River	Boston	296,700	09/16/76
"	Cochituate	Natick	125,000	08/26/76
"	Ellis Brett Pond*	Brockton	106,500	01/05/77
"	Lower Mystic	Boston	320,000	11/21/77
"	Nutting*	Billerica	166,211	05/12/77
Michigan	Lansing*	Mason	1,255,957	04/13/76
"	Reeds	E. Grand Rapids	903,452	06/29/77
Minnesota	Long*	Arden Hills	1,296,715	12/18/76
"	Phalen	Minneapolis-St. Paul	575,683	01/18/77
"	Albert Lea*	Albert Lea	302,800	01/08/76
"	Clear	Waseca	269,075	09/27/76
"	Chain of Lakes	Minneapolis	179,000	02/13/76
"	Hyland	Maple Run	161,198	08/10/76
"	Penn*	Bloomington	87,900	03/21/77
Missouri	Vandalia*	Vandalia	350,000	02/06/78
"	Finger	Columbia	50,000	02/16/78
"	Rothwell*	Moberly	45,000	02/16/78
New York	Ronkonkoma	Islip	334,048	09/29/76
"	Delaware Park*	Buffalo	141,500	02/17/76
"	Tivoli	Albany	121,500	04/11/77
"	Hampton Manor*	E. Greenbush	50,000	04/09/76
"	Collins Park*	Scotia	79,355	01/08/76
"	Steinmetz*	Schenectady	36,680	12/30/76
"	Buckingham*	Albany	23,250	04/09/76
"	Washington Park*	"	"	"
"	59th Street Pond*	New York City	325,020	05/12/77
North Carolina	Mystic*	Lake Lure	21,080	09/30/77
Oklahoma	Pauls Valley	Pauls Valley	300,000	08/16/77
Oregon	Commonwealth*	Beaverton	53,200	12/21/76

(continued)

TABLE 2 (continued)

State	Lake	Grantee/Location	Amount	Date of First Award
South Dakota	Kampeska	Watertown	$ 67,000	08/25/76
"	Swann	Viborg	39,500	12/07/76
"	Oakwood	Brookings	26,500	08/10/76
"	Cochrane	"	9,906	01/08/76
Texas	McQueeney*	Sequin	120,000	11/15/77
Vermont	Bomoseen	Castleton	74,640	06/15/77
Virginia	Rivanna	Charlottesville	39,728	03/23/77
Washington	Liberty*	Spokane	577,975	02/07/77
"	Long*	Port Orchard	355,970	01/08/76
"	Moses	Moses	124,675	08/31/76
"	Spada/Chaplain	Everett	99,000	01/08/76
"	Medical	Medical Lake	128,217	12/21/76
"	Vancouver*	Vancouver	25,000	08/23/76
"	Sacajawea*	Longview	1,717,562	01/06/78
Wisconsin	Little Muskego*	Wheatland	995,000	08/10/76
"	Half Moon	Eau Claire	371,500	01/18/76
"	Lilly*	Wheatland	273,000	01/08/76
"	Mirror/Shadow	Waupaca	215,000	"
"	Noquebay	Marinette	245,000	08/23/76
"	Henry*	Blair	220,000	12/21/76
"	White Clay	Madison	107,200	01/08/76

* Notes projects which include dredging or sediment treatment.

the decision to seek permission from the Office of Management and Budget to develop a regulation to establish policies and procedures for grant assistance to the States in conjunction with Section 314 of the Clean Water Act. Permission was given and regulations concerning grants for the restoration and protection of lakes in the United States are now being developed.

INITIAL PROGRAM GUIDANCE

At the outset of the Clean Lakes Program in 1975, there was no assurance of its viability. It was a relatively small (first year funding was $4 million) program with what appeared to be limited and somewhat regionalized support. Therefore, instead of adopting new regulations and gearing up for a new program, it was decided to operate within the existing framework of EPA grant regulations. As indicated previously, the research and demonstration section of the Act, Section 104 (h), was selected as the funding vehicle. It followed from this that the Clean Lakes granting practices would have to comply with the research and demonstration grant regulations. This operating procedure was adopted, and general guidelines were formulated for the preparation of lake restoration grant applications (USEPA, 1976).

The 1976 Guidance Document specified that EPA was most interested in funding lake restoration projects which:

1. Employed methods to restore lake water quality by eliminating or otherwise restricting the input of non-point source waste influents to the lake or that part of its drainage basin where a contributing pollutional effect on the lake water quality could be demonstrated and;

2. Used in-lake techniques to remove or inactivate undesirable pollutants, including nutrients, after steps were taken in the watershed to reduce their rapid re-accumulation.

The following statement from the Guidance Document provides further evidence that the Clean Lakes Program would stress watershed, non-point source pollutant control technologies over in-lake treatment techniques. "Reducing or eliminating the sources of waste loading may be the only restorative measure that is needed to achieve the desired level of improvement of certain lakes" (USEPA, 1976). The statement has a strong basis if one considers the percentage of various pollutants estimated to be derived from non-point sources (Table 3). It also is clear that the program is directed toward the problem sources most frequently identified in Table 1 by Ketelle and Uttormark in 1971.

The Guidance Document further states that Clean Lakes funds provided under Section 314 will not be used to control point source waste where the issuance of permits (according to Section 402 of the Act) or the construction of wastewater treatment facilities (according to Section 201 of the Act) will alleviate the causes of pollution. The following types of projects are listed as examples of those eligible for funding:

TABLE 3. ESTIMATED PERCENTAGE OF TOTAL POLLUTANT DISCHARGES WHICH ARE DERIVED FROM NON-POINT SOURCES (from Freeman, 1978)

Pollutant	Non-Point Sources as Percentages of Total Discharge
Suspended Solids	92
Fecal Coliform	98
Total Nitrates	79
Total Phosphates	53
BOD	37

- Lake drawdown and lake bed consolidation;
- Use of flocculants to precipitate nutrients;
- Dredging or covering of sediments;
- Dilution and displacement of pollutants with good quality water;
- Purchase or long-term leasing of land easements to establish buffer zones for the control of runoff and its effects;
- Diversion of nutrients and sediments from lakes;
- Improving agricultural and other land use practices to keep soil and nutrients in the watershed.

Projects considered to be ineligible for funding under the program are described in a later section of this paper under Federal Funding Restrictions For Phase 1 and Phase 2 grants.

CURRENT LAKE RESTORATION PROGRAM POLICY

There is little doubt that the intent and policies of the program were designed to control and reduce non-point source pollutants. The program was tailored to complement the Section 208 Areawide Waste Management Planning Program. In this respect, by late 1977, several States began making requests for Federal assistance to develop technically sound lake restoration proposals. However, the initial EPA program guidelines specifically prohibited Federal funds from being used for that purpose. The funds were for use in actual lake restoration implementations only. There was a good deal of confusion over the 50% Federal/50% grantee cost-sharing policy established by EPA for the Clean Lakes Program and the customary 95% Federal/5% grantee cost-sharing policy for research and demonstration grants administered under the same section of the law [104(h)]. Another point of difficulty arose from the fact that Section 314 of the Act stated that grants under this program would be made only to the States. No other organizations or governmental levels were mentioned. However, Section 104 permits grants to go to State agencies, local governmental units, or any group elected by and representing the general public in areas of control or regulatory authority over a publicly owned freshwater lake. Inconsistencies between the Section 314 mandate and the Section 104 funding vehicle were making the increasingly complex Clean Lakes Program difficult to manage. While the program had been initiated as a demonstration program, there was considerable pressure from States and EPA Regional offices to convert it to an operational program. All of these considerations led to the conclusion that a new governing regulation would be necessary to establish uniform policies and procedures for administering the Clean Lakes -- Lake Restoration Program. The new regulatory/policy language is currently being formulated with the following goals in mind:

1. To present a uniform set of requirements to define eligible grant applicants and specify the types of projects that would qualify for assistance under the program;
2. To provide grant applicants with explicit information requirements to be included in all applications;
3. To assist States in complying with the requirement to prioritize their lakes for restorative action;
4. To define the administrative procedures used by EPA to receive and review proposals and prepare funding recommendations; and
5. To define Federal funding for lake restorative and pre-restorative activities.

The following five sections of this paper and Appendix A draw heavily from the draft regulation document prepared by the Criteria and Standards Division of the Environmental Protection Agency. It should be understood that this information is currently in draft form and subject to change. A proposed regulation will be made available for a 60-day public comment period. After that time the final regulation will be developed and promulgated.

Eligibility

Freshwater lake under the draft regulation refers to any inland pond, reservoir, impoundment, or other similar body of water that exhibits no marine water intrusion as a result of oceanic or tidal activities. The water body must have public recreational value and, therefore, public access. Reservoirs used solely for drinking water supplies are prohibited from receiving funds under this program. In accordance with Section 314 of the Act, the new regulation would allow grants to be made only to States. However, it would be permissible for the States to enter into interagency agreements with sub-state entities (county, township, municipality, city, watershed district, etc.) to perform various tasks included in the scope of work of specific grants. In other words, sub-state entities could act as sub-contractors to the States. All such sub-state interagency agreements would be reviewed by the EPA Regional Administrator and subject to his approval if the sum of the agreement exceeded $100,000.

The types of projects eligible for funding will remain essentially the same under the new regulation as they had been under the initial program guidance document. One significant exception will be mentioned under the following section on types of grant assistance and information requirements and under the section defining funding for restorative and pre-restorative activitiy.

Types of Grant Assistance and Information Requirements for Application

Perhaps the single most significant deviation from the initial program operating procedure in the new Clean Lakes policies is the one which deals with pre-implementation diagnostic-feasibility studies. Support for pre-implementation studies was specifically prohibited in the initial program guidance document. It has become clear that many States require financial and technical assistance in these endeavors. The new regulation authorizes pre-implementation studies to identify and evaluate lake characteristics and to develop a restoration or water quality protection plan as a part of the Clean Lakes Program. This type of support is called a Phase 1 matching grant. The Federal government will provide up to $100,000 on a 70% Federal/30% grantee cost-sharing basis for Phase 1 grants. Requirements for the diagnostic-feasibility studies are included in this paper as Appendix A.

Actual implementation of pollution control measures or in-lake restoration methods and procedures are to be supported by Phase 2 matching grants. These grants are the equivalent of those formally issued under the initial guidance document. The Federal/state cost-sharing percentages for Phase 2 grants are proposed to remain at the 50% Federal/50% grantee level. The Agency has adopted the position that this funding formula will provide sufficient local commitment to assure optimization of project implementation and a concern for proper maintenance of the project after it is completed. Phase 2 grant proposals must contain all of the information specified in Appendix A to receive funding consideration. This represents a significant tightening of the requirements for funding an implementation project.

State Prioritization of Lakes

Section 314 of the Water Pollution Control Act Amendments requires States to identify and classify their lakes according to eutrophic conditions. The draft regulation specifies that a State must accomplish this by January 1, 1982, or become ineligible to receive Federal funds under the Clean Lakes Program. Phase 1 grant funds may be used by States in order to comply with this requirement. The method of classification and prioritization is left to the discretion of individual states.

Application, Review and Evaluation

Annual appropriations for the Clean Lakes Program have never exceeded $15 million. The Office of Water Planning and Standards has taken the position that there is no equitable way to distribute such a small amount of money among the States and still maintain a technically and administratively sound program. Therefore, the policy will be to continue to review grant applications as they are received and to fund them, project by project based on technical merit.

Phase 1 and Phase 2 grant applications will be considered separately, but both will be evaluated according to the procedures outlined in Figure 1 and in terms of:

- Technical feasibility and the estimated improvement in lake water quality as determined through information compiled by the grant applicant;
- The anticipated positive changes in the overall lake ecosystem with regard to sediment, nutrient, and pollutant loading and the subsequent effects on lake biomass as a result of project implementation;
- The degree, nature and sufficiency of unencumbered public access to the lake;
- The size of the population surrounding the lake which would realize recreational benefits as a result of the project;
- Other relatively clean public owned freshwater lakes which may serve the same population adequately;
- Whether the restoration project would disproportionaly benefit private land owners adjacent to the lake as compared to those somewhat removed from the shoreline;
- The reasonableness of proposal costs in consideration of the various proposed tasks, likelihood of project success and the projected benefits;
- The prospect of dealing satisfactorily with adverse environmental impacts resulting from the proposed course of action;
- The level of state priority for a particular grant application; and
- The recommendation of the review panel members, including those of technical experts from the university community.

Within 90 days of the receipt of an application it will be either funded, rejected or returned to the applicant for additional information or due to lack of funds. In the latter two cases, applications may be resubmitted when the issues identified during review have been resolved or when additional funds become available.

Federal Funding Restrictions for Phase 1 and Phase 2 Grants

Each State may apply for up to $100,000 in Phase 1 pre-implementation, Federal grant funds (on a 70% Federal/30% grantee cost sharing basis) to assist them in developing complete, technically sound programs for implementing lake restorative procedures. An application may address more than one lake. These funds will come from the annual Clean Lakes appropriation and may amount to as much as 20% of that appropriation, but will in no event exceed $5 million per year. Each grant will be approved for a period not to exceed 2 years.

Further restrictions require that the Regional Administrator will determine that pollution control measures in the lake watershed have been taken in accordance with Section 201 (Wastewater Treatment Plant Construction Program), approved 208 planning (Areawide Waste Treatment Planning Program), and Section 402 (Pollutant Discharge Permitting Program). These activities must be completed or progressing according to the time schedule of an approved plan, or discharge permit schedule, by the time a lake restoration (Phase 2) project is completed. Unlike the Phase 1 grants, Phase 2 grants have no upper level funding limits. As per the initial 1976 guidelines, Clean Lakes funds may not be used to support projects to control point source discharges (Section 402) or for planning and construction of wastewater treatment facilities (Section 201) where those programs will resolve the pollution problems. Certain other types of projects are similarly prohibited. These include projects using chiefly palliative methods which treat symptoms of pollution rather than attempt to eliminate the source of the problem, e.g. harvest of aquatic vegetation, herbicide treatment, and maintenance of lake aeration devices. Clean Lakes funds may not be used for desalination procedures or for the purchase or long-term leasing of land to provide unencumbered public access to a lake.

The Environmental Protection Agency presently is proposing revised Water Quality Management Regulations to conform to the recently developed extended management strategy for Water Quality Management, FY-1979-83. These regulations set forth an increased emphasis on State/EPA Agreements. The initial results of Clean Lakes Projects clearly show that this program can be the basis for effectively implementing pollution control practices that improve dramatically the water quality of both tributary streams and lakes. State and local communities have shown a great deal of interest in these results and appear impressed with the potential of this program to assist them with water quality improvement. Thus, the gradual development of lake restoration policy has served as an asset. It is anticipated that the newly formulated policies and procedures of the Clean Lakes Program will facilitate its implementation.

SUMMARY

-- This paper provides a statement on the type and extent of water quality problems associated with freshwater lakes in the United States.

-- It traces the historical course of the Federal government's involvement in water pollution control activities.

-- It explains the role of EPA's Office of Water Planning and Standards in administering the Clean Lakes Program and the role of the Office of Research and Development in evaluating the effectiveness of various lake restoration techniques.

-- It describes the initial policies of the program and the evolution to their current state.

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APPENDIX A

GUIDANCE FOR DIAGNOSTIC FEASIBILITY STUDIES

Proposals for Phase 1 Clean Lakes grants must include the following information in their scope of work for all Clean Lakes applications.

Diagnostic Study Requirements

1. An identification of the waters to be restored or studied, including their name, state in which located, location within the state, area, maximum depth, average depth, detention time, the general relationship with associated upstream and downstream waters.

2. A geological description of the drainage basin including soil types and soil loss to stream courses that are tributary to the lake.

3. A description of the public access to the lake.

4. A summary of historical lake uses including recreational lake uses through the present time and how these may have changed through the years because of water quality degradation. A statement must be made regarding the water use of this lake compared to other lakes within a 50 mile radius.

5. An itemized inventory of known point source domestic or industrial pollutional discharges affecting or which have affected lake water quality over the past 5 years and the abatement actions that have been taken, are in progress, or are contemplated within a specified time period.

6. A description of the land use in the lake watershed with an indication of what percentage of the watershed each encumbers.

7. A discussion of non-point pollution loading derived from each identified land use category.

8. A discussion and analysis of current (within one year from the date of application) baseline limnological data. Such data must include the present trophic condition of the water body as well as its surface area, maximum depth, average depth, hydraulic residence time, area of the watershed draining to the lake, and the physical, chemical, and biological impact of important tributaries. Bathometric maps should be provided and where dredging is part of the restoration plan, representative bottom sediment core samples and analyses for nutrient content, heavy metals and persistent organic chemicals must be provided. An assessment of nitrogen and phosphorus inflows and outflows associated with the lake and a hydraulic budget including groundwater flow must be included. Vertical temperature and dissolved oxygen data are essential and it must be determined if the hypolimnion becomes anaerobic and if so, for how long and over what extent of the bottom. The extent of algal blooms

and the predominent algal genera must be discussed. The portion of the shore line and bottom that is impacted by vascular plants (submerged, floating, or emersed higher aquatic vegetation) must be estimated and that estimation must include an identification of the predominant genera. There must be an identification and discussion of major biological resources in the lake, such as fish populations and how they might be expected to change.

Feasibility Study Requirements

1. An identification and discussion of the alternatives for pollution control or lake restorative action considered; and an identification and justification of a selected alternative when considering all alternatives including a discussion of expected water quality improvement, technical feasibility, and estimated costs that are attached to each alternative.

2. A discussion of the particular public benefits expected to result from implementation of the project, including new public water uses that may be associated with the enhanced water quality.

3. A proposed monitoring or investigative program with schedule for evaluation. Based on the information supplied by the applicant and the technical evaluation of the proposal, a detailed monitoring program to evaluate project effectiveness during and up to one year following the completion of project implementation will be established for each approved project and will be specified in any grant agreements supporting implementation or restorative procedures. An additional guidance for monitoring methods and procedures will be provided at the time of grant award.

4. A proposed milestone work schedule for project completion with a proposed budget and payment schedule relating to progress milestones.

5. A detailed description of how matching funds will be obtained to support the non-federal funding requirements for the proposed project.

6. A description of pertinent relationships of the proposed project to local, state and federal pollution control programs such as the Section 201 Construction Grants Program, the Section 208 Areawide Wastewater Management Program, the Soil Conservation Service Programs as described by PL 83-566, the Department of Housing and Urban Development Programs, and any other local, state, and federal programs which may bear a relationship to the proposed project.

Environmental Evaluation

An environmental evaluation to consider all potential adverse impacts of the project must be included with the application. The purpose of this document is to identify and thus minimize the effects of any adverse impacts which may be associated with the project.

SEDIMENTS AND SEDIMENT DISTURBANCE DURING DREDGING

John F. Sustar
U.S. Army Engineer District, San Francisco
San Francisco, California 90105

INTRODUCTION

During the past ten years numerous studies have been conducted in the field to quantify the disturbance during dredging operations. The results have generated wide ranges of data. The conclusions, however, do state that improving the dredging efficiency will decrease impacts or possible impacts of dredging operations. Despite extensive studies in San Francisco Bay, significant chemical and physical stresses on the biological communities have not been identified. The conclusion on improving dredging efficiency, however, is still valid in terms of not only minimizing the disturbance to the biological system, but also minimizing the investment in maintaining channels. Minimizing disturbance is also important where known isolated contamination is being removed.

Studies of sediment disturbance during dredging have been continuously modified not only to locate the disturbance but to develop handles on identifying the parameters that control the disturbance. Figure 1 shows schematic diagrams of the three most common dredging methods in San Francisco Bay—the trailing suction hopper dredge, the hydraulic cutterhead dredge and the clamshell dredge. The figure also shows the generally recognized areas of disturbance.

Yagi et al. (1976) looked at both the hydraulic cutterhead dredge and the grab bucket (clamshell) dredge. For the hydraulic cutterhead, they concluded that: "The accumulative ratio of short-absorbed soil, however, decreases with the increase of the number of cutter swings. On the other hand, the turbidity has the general trend to increase in opposition to the above mentioned trend." Their work dealt primarily with the swing speed and the sediment thickness. Shape of bottom was also concluded to be an important parameter. The sediment type was only mentioned in terms of the gradient of the vertical turbidity distribution. With the grab bucket, turbidity was found to be largely dependent on the type of sediment. The closed bucket, which reduces the interaction of localized eddies with the sediment interface in the bucket, did decrease the loading of the water column.

Huston (1976) concluded that: "Techniques for reducing dredge-induced turbidity consist principally of good dredging procedures and the proper use of existing dredging equipment. Some of the items discussed included disturb-

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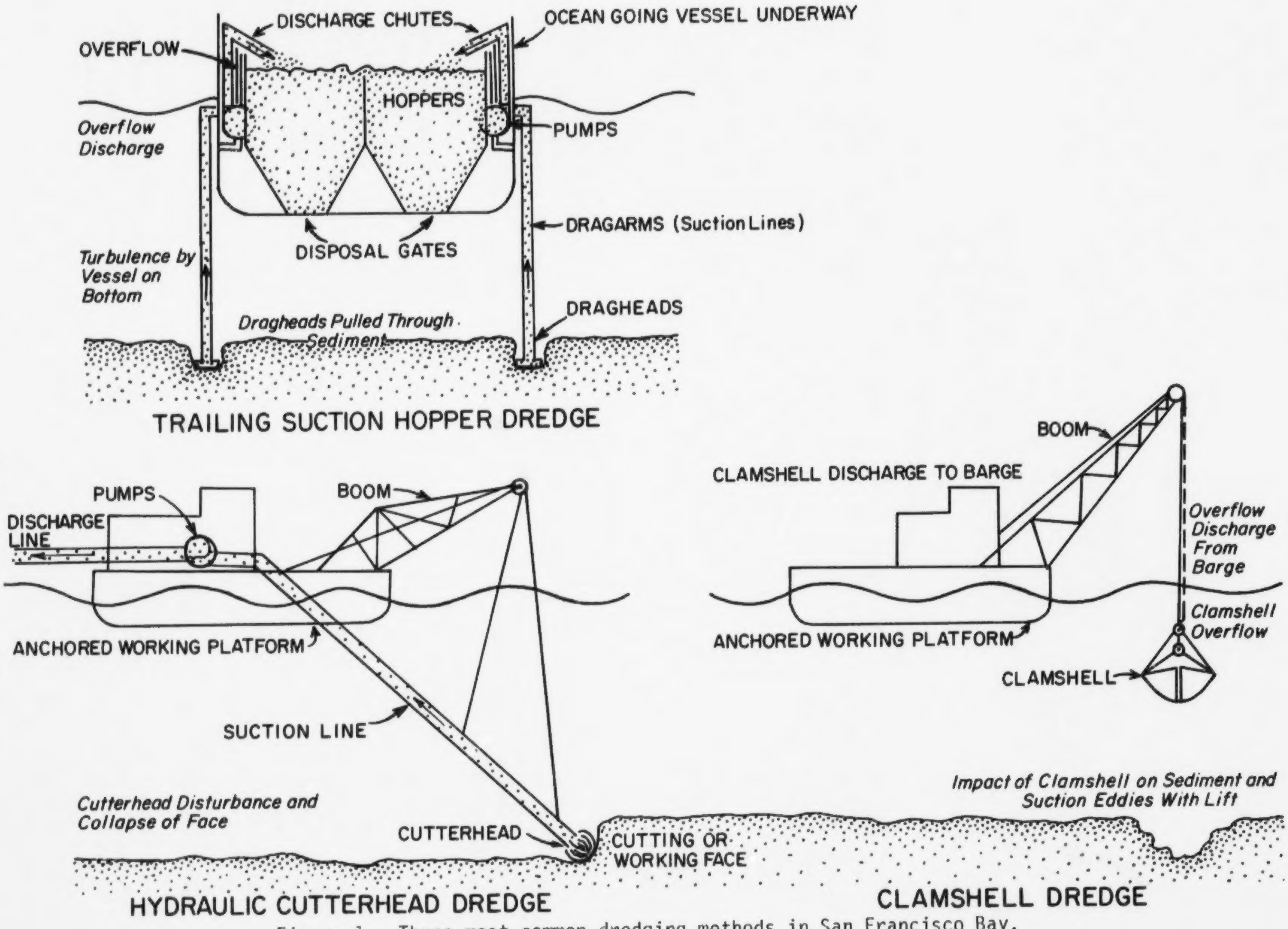

DISCHARGE CHUTES
OCEAN GOING VESSEL UNDERWAY
OVERFLOW
HOPPERS
Overflow Discharge
PUMPS
DRAGARMS (Suction Lines)
DISPOSAL GATES
Turbulence by Vessel on Bottom
DRAGHEADS
Dragheads Pulled Through Sediment
TRAILING SUCTION HOPPER DREDGE
BOOM
CLAMSHELL DISCHARGE TO BARGE
PUMPS
BOOM
DISCHARGE LINE
Overflow Discharge From Barge
Clamshell Overflow
ANCHORED WORKING PLATFORM
ANCHORED WORKING PLATFORM
CLAMSHELL
SUCTION LINE
Impact of Clamshell on Sediment and Suction Eddies With Lift
Cutterhead Disturbance and Collapse of Face
CUTTERHEAD
CUTTING OR WORKING FACE
HYDRAULIC CUTTERHEAD DREDGE
CLAMSHELL DREDGE

Figure 1. Three most common dredging methods in San Francisco Bay.

ances by the vessels themselves including the tenders, the inefficiency of cleanup operations and the effectiveness or skills of the personnel. His work did not discuss the relationship of the equipment and the type of sediment in disturbing both the bottom and the water column, although proper use of equipment does imply the selection of the type of equipment for a given project. With the spectrum of dredging operations and available equipment, this selection process is not necessarily controlled by the optimization of the operation efficiency.

The work by Weschler et al. (1977) does employ a settling jar to evaluate sediment characteristics for use in a numerical model. The mention, however, does not explain what actually occurs in the system.

Studies by Johnson et al. (1975) showed the large scale heterogeneous loading characteristics of both the hopper dredge and the clamshell with dump scow. With the operation (elevation changes) of the dragheads on the hopper dredge, major differences were observed.

Sustar et al. (1976) identified initial dispersion patterns related to the type of sediment and the dredging equipment. The major portion of their studies with the dredging operation concentrated on locating the plume and describing the plume vertically and horizontally by percent transmission and suspended solids loading. Although patterns were discernible, variation of data was very large. Studies were equipment oriented and project oriented.

The type of sediment being dredged has generally been ignored in terms of interpreting the disturbances that occur in the dredging area. Based on previous studies in San Francisco Bay on the release pattern during open water disposal, dredging operations in the Bay were evaluated in terms of the type of sediment and the efficiency of the operation. An interpretation of the bottom and water column disturbance using actual dredging operations and sediment testing is given.

DREDGING EFFICIENCY

If the basic premise (increased efficiency decreases disturbance) were reversed, a greater disturbance during the dredging should reduce the efficiency of the operation. To test this, three dredging periods in Mare Island using the trailing suction hopper dredge HARDING were evaluated. Figure 2 shows the location of Mare Island Strait.

The shoaling in Mare Island Strait is silty-clay. The October-November 1975 dredging period represented a shoal formed over a several month period with sediments recirculating in the northern portion of the Bay. The second period was February-March 1976. The shoaling generally represents new sediments entering the system with outflows from the Sacramento-San Joaquin Delta.

The third period was July-August 1978. Following an extended drought, flood flow brought new sediments from the Delta. The delayed dredging from the usual February-March cycle probably developed with progressive movement of sediments through the Delta and Carquinez Strait with each storm following the drought. The total dredging cycles for each of the periods were 606, 430 and

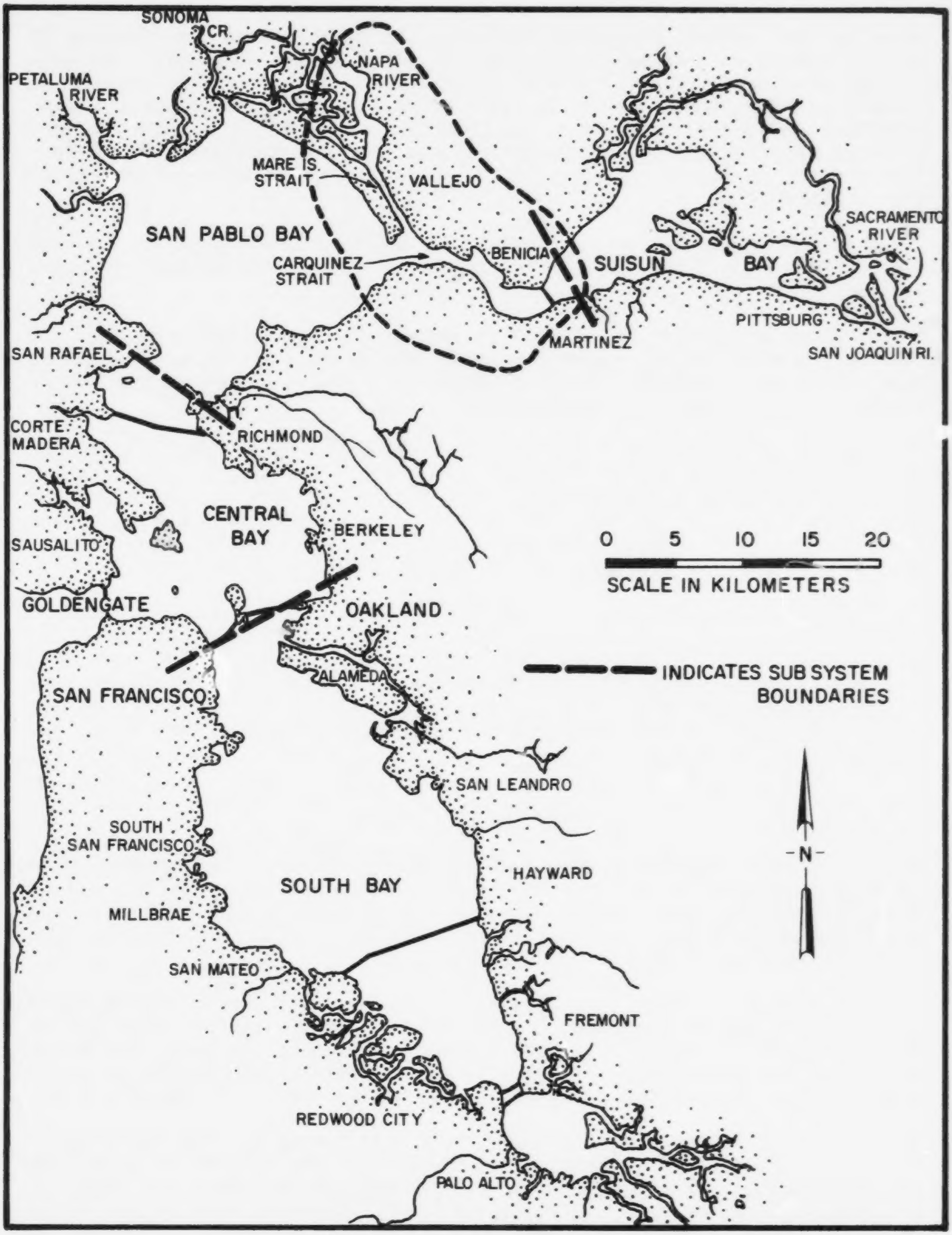

SONOMA CR.
PETALUMA RIVER
NAPA RIVER
MARE IS. STRAIT
VALLEJO
SAN PABLO BAY
CARQUINEZ STRAIT
BENICIA
SUISUN BAY
SACRAMENTO RIVER
PITTSBURG
MARTINEZ
SAN JOAQUIN RI.
SAN RAFAEL
RICHMOND
CORTE MADERA
CENTRAL BAY
BERKELEY
SAUSALITO
GOLDENGATE
OAKLAND
ALAMEDA
SAN FRANCISCO
SAN LEANDRO
SOUTH SAN FRANCISCO
SOUTH BAY
HAYWARD
MILLBRAE
SAN MATEO
FREMONT
REDWOOD CITY
PALO ALTO
0 5 10 15 20
SCALE IN KILOMETERS
INDICATES SUB SYSTEM BOUNDARIES
N

Figure 2. Mare Island Strait.

396, respectively. Shoaling patterns in Mare Island Strait are very definite with buildups concentrating in three to four areas.

From the vessel's records, the cubic yards per pumping minute was obtained for each cycle using the load meter and the pumping time. The data were averaged in ten cycle groups to reduce the variability between successive cycles. The variability between cycles results from the heterogeneous nature of the bottom and variations in dredging location with movement of vessels and day/night time dredging. The average efficiency and plus and minus one standard deviation are shown in Figures 3 through 5.

All three periods show a decrease in loading efficiency with time. The first and second periods show definite cycles of decreasing efficiency indicating a work pattern of removing one shoal at a time. The type of equipment (trailing drags) working the shoal of silty-clay causes both trenching of the shoal and disturbance with mixing of water in the shoal. After a number of passes through a shoal, the dragheads will cross previous trenches resulting in the intermittent variation of suction at the intake, increased water in the slurry and subsequent reduction in the density of the pumped sediment. The decreased loading efficiency of each of the discernible cycles (low ten cycle average divided by preceding high ten cycle average) is as follows:

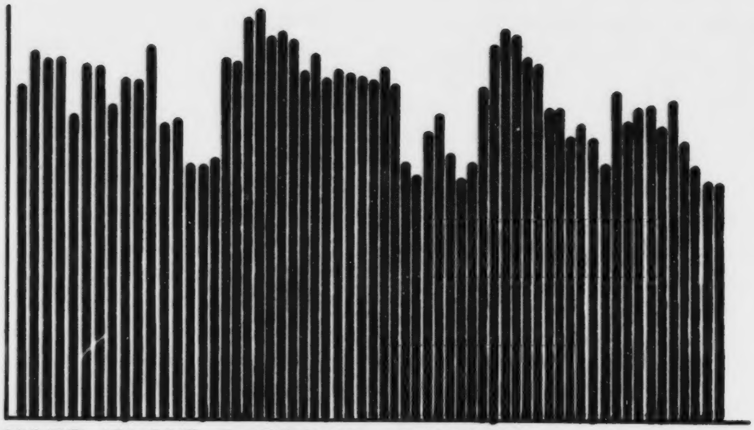
	Oct-Nov 75	Feb-Mar 76	Jul-Aug 78
1st cycle	72%	42%	55%
2nd cycle	59%	57%	---
3rd cycle	62%	81%	---

The decrease is based on an "average optimum" condition with the initial dredging of an undisturbed area.

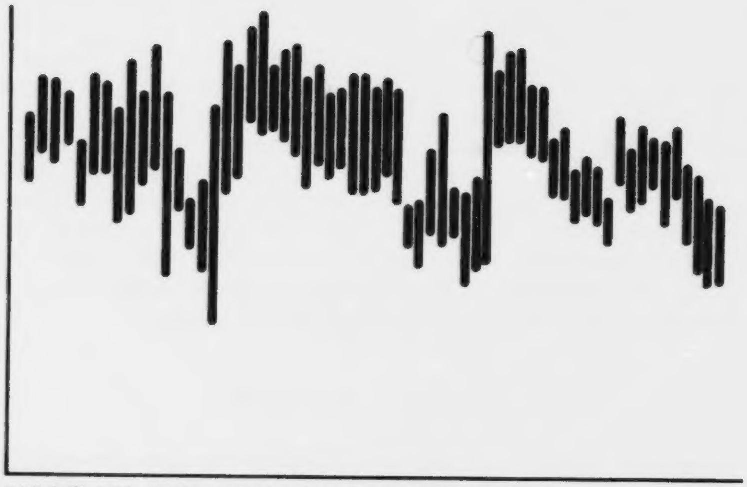
Although patterns of decreasing efficiency of pumping are evident, the analysis also shows some increases in efficiency. As the shoals form with time, different densities occur in the bottom. Assuming optimal operation of the vessel including the control of the dragheads and the speed of the vessel relative to the bottom, pumping efficiency is dependent on the density of the sediment. As the bottom sediments are disturbed, increasing the water content, an initial increase in pumping efficiency will occur until an optimum pumping density is reached. As the disturbance continues, the density continues to decrease, decreasing the pumping efficiency.

SEDIMENT RELATIONSHIPS

Three types of sediments were collected in San Francisco Bay to examine qualitatively the comparative degree of disturbance with resulting bottom configuration (trenching) and suspended particulate. The classification of the sediments is as follows:

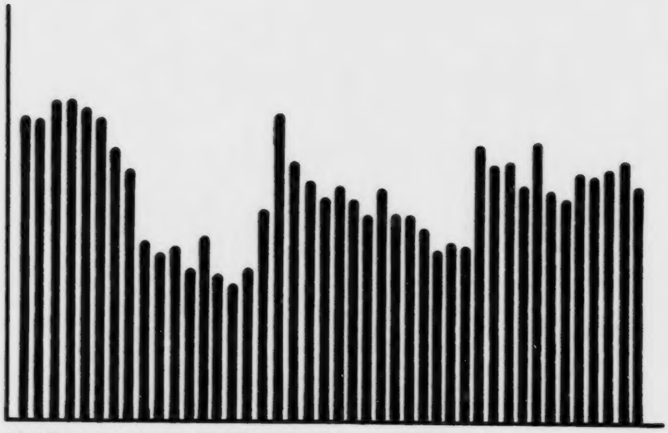


MARE ISLAND DREDGING - HARDING
OCT - NOV 1975 YD3/ MIN VS TIME
AVERAGE 18 CYCLES
HIGH 117 LOW 68

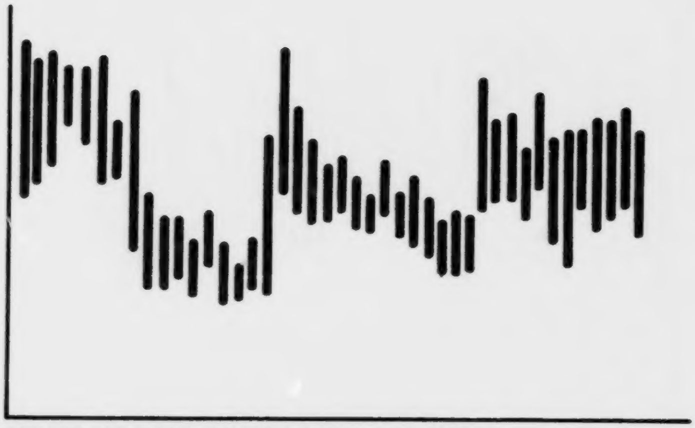


MARE ISLAND DREDGING - HARDING
OCT -NOV 1975 YD3/ MIN VS TIME
TWO STANDARD DEVIATIONS 18 CYCLES
MAXIMUM 32 MINIMUM 5

Figure 3.

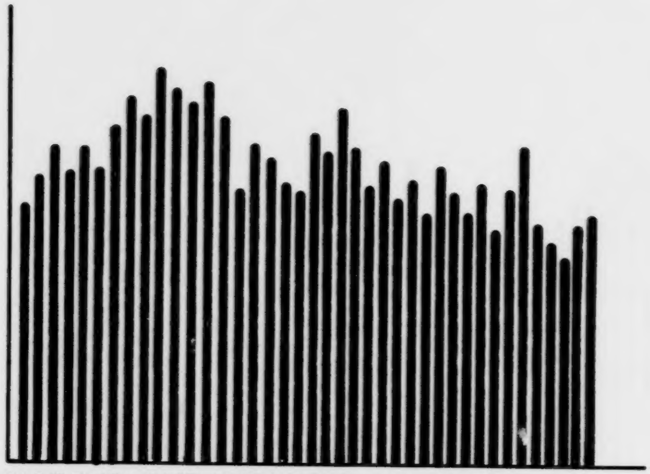


MARE ISLAND DREDGING – HARDING
FEB – MAR 1976 YD3 / MIN VS TIME
AVERAGE 18 CYCLES
HIGH 92 LOW 39

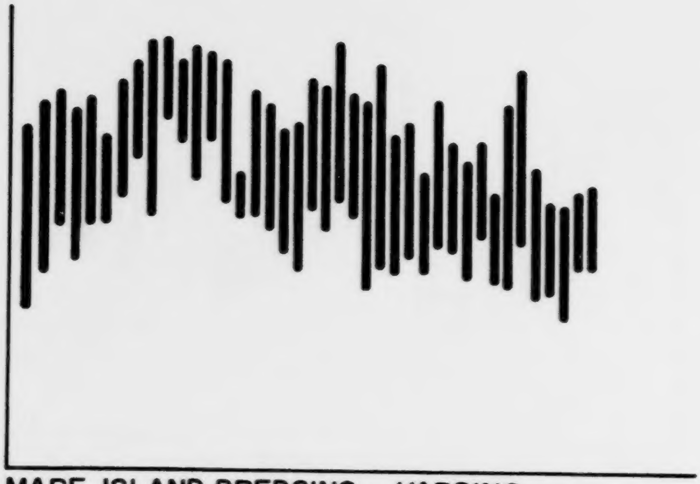


MARE ISLAND DREDGING – HARDING
FEB – MAR 1976 YD3 / MIN VS TIME
TWO STANDARD DEVIATIONS 18 CYCLES
MAXIMUM 22 MINIMUM 5

Figure 4.



MARE ISLAND DREDGING - HARDING
JUL-AUG 1978 YD3/ MIN VS TIME
AVERAGE 18 CYCLES
HIGH 113 LOW 59



MARE ISLAND DREDGING - HARDING
JUL-AUG 1978 YD3/ MIN VS TIME
TWO STANDARD DEVIATIONS 18 CYCLES
MAXIMUM 26 MINIMUM 6

Figure 5.

	Classification	Liquid Limit	Plastic Limit	PI
1.	Silty Sand (SM)	31	26	5
2.	Sand (SP)	--	--	--
3.	Clay (CH)	99	38	61

The gradation curves are shown in Figure 6.

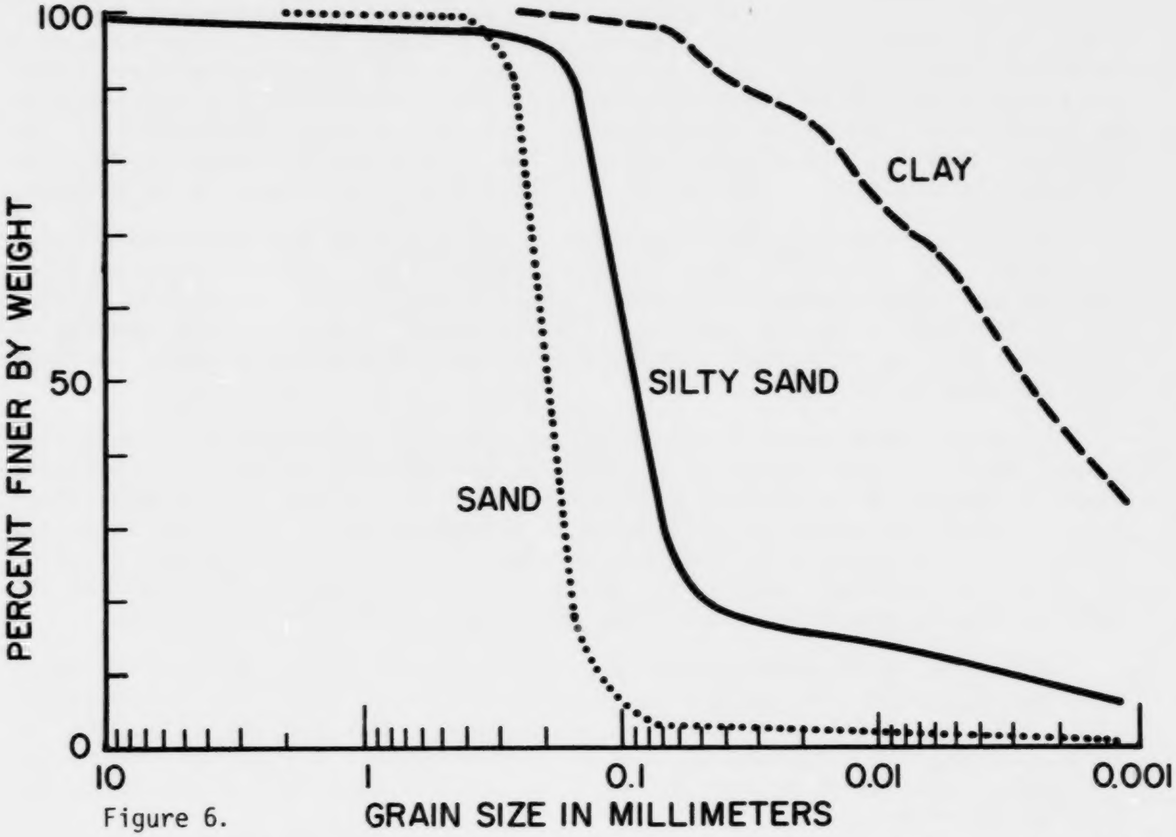


Figure 6.

The sediments were placed in clear 30 cm x 16 cm containers with nine centimeters of salt water over the sediments. A standard disturbance was made in each container using a nine millimeter diameter rod. The rod was pulled along the same line in groups of five pulls. Uniform speed and pressure were maintained.

The clay, sample 3, produced the best defined trench with vertical side walls. The width of the trench (15 cm) was not significantly changed with increased disturbances along the same line. The silty sand, sample 1, also developed a trench. Vertical walls were present, but were not as well defined as with the clay. The width of the trench (25 cm) increased with increasing disturbance. The sand, sample 2, developed a V-shaped trench, 30 cm at the top.

Sand produced very little turbidity when comparing the suspended particulate to a photo density scale (visual observation). The silty sand almost immediately produced turbidity with very little increase in turbidity with additional disturbance. The clay produced successive increase in turbidity with each additional disturbance. The maximum turbidity with the clay was greater than with the silty sand.

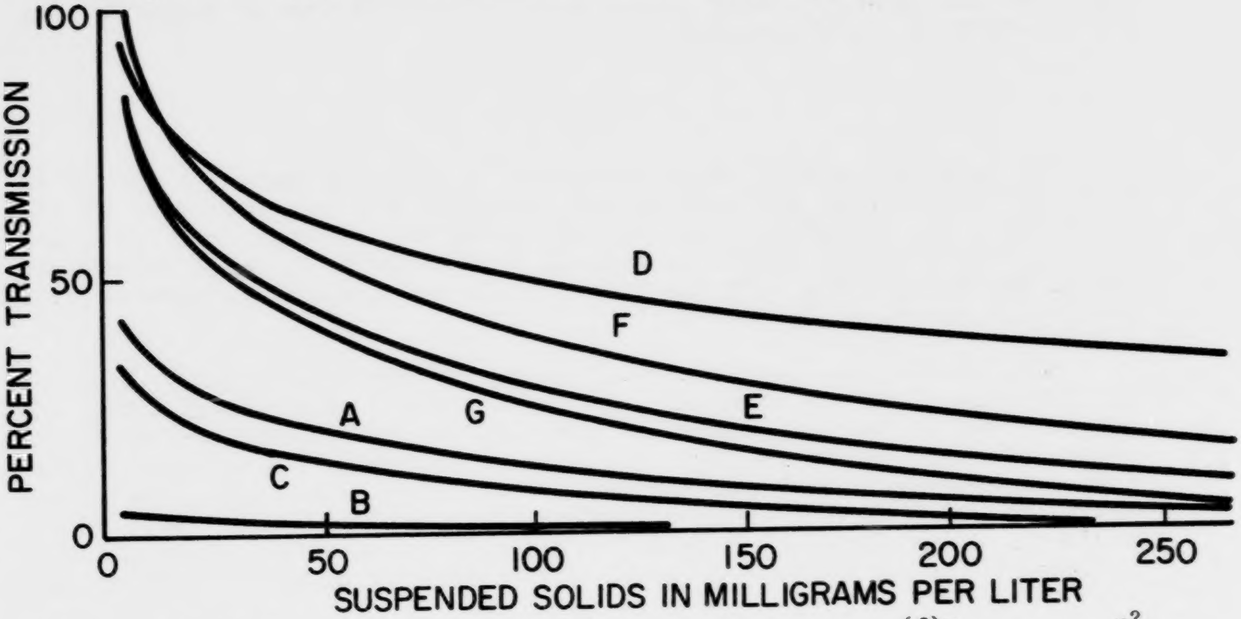
The differences in the trenches can be explained by the type of sediment irrespective of the type of disturbance. Following a disturbance the sand comes to an angle of repose, resulting in a V-shaped trench. The silty sand maintains some vertical wall because it is a low cohesive sediment. The increased width of the trench results from the susceptibility of the sediment to erode with successive disturbances. As the cohesive properties of the sediment increase, the direct action by the disturbing instrument is required to move the sediment. The narrow width of the clay trench is an example.

Within San Francisco Bay, the type of sediment does influence the pumping efficiency when comparing Mare Island Strait (clay), Richmond Harbor (silt) and the San Francisco Bay (fine sand). Mare Island Strait sediment is conducive to the most efficient pumping. The San Francisco Bar sediment results in the least pumping efficiency. Pumping efficiency is defined by cubic yards of load per pumping minute.

In Mare Island Strait, the sediments are less susceptible to mixing with water; that is, more energy or disturbance is required to decrease the density. It should be noted that efforts are made to control the dredging lines (i.e., reduce the crossing of trenches). Dredging cycles have been conducted in Mare Island Strait with a hopper dredge in which no overflow was required to attain an economic load. This means that the sediments were near optimum pumping density and excellent control of dredging lines was maintained.

The silts in Richmond Harbor are easily disturbed, resulting in a sediment density below the optimum pumping density. The sands on the San Francisco Bar are independent of the disturbance. The pumping efficiency is controlled by the efficiency of the pumps rather than the disturbance of the sediment. The speed of the dredge over the bottom largely influences the loading characterization.

The same dredging operations in San Francisco Bay illustrate the levels of suspended particulates during dredging. Greater turbidity (reduced light transmission) results from clays because of the increased number of particles in suspension per unit weight as compared with silts. The level of clay loading (weight suspended solids) is dependent on the duration of disturbance (addition and mixing of water to decrease the cohesive properties) and the flocculation rate. The flocculation rate is dependent on the concentration of suspended solids, salinity of the system and collision of particles (disturbance in the operation, i.e., overflow, and in the water column, i.e., passage of the vessel). With Mare Island Strait sediments, the non-dispersed grain size is about 20 micron. Without additional disturbance, the clays within the hydraulic regime will react as silts. Although the level of suspended solids in the water column between clays and silts may be similar, percent transmission will be dramatically different. Figure 7 represents a logarithmic regression analysis of seven sets of data. The data present in situ percent transmission readings, using a ten centimeter light pattern, compared with suspended solids analysis of samples pumped from adjacent to the transmissometer. The analysis does fit the above description. The clays of Mare Island Strait generate the highest turbidity (decreased light transmission) as compared to the coarser silts in other areas such as Richmond Harbor and Alameda Naval Air Station. Within the three regression curves of Mare Island Strait, the lowest of the curves is at a time of lower salinity with freshwater flows coming from the Sacramento-San Joaquin Delta. The lower salinity of both the sediment and the water decreases the rate of flocculation and increases the time of sediments in suspension. Dredging with stratified water conditions in Mare Island Strait introduced saline water from the lower water column into the freshwater in the upper water column. The results were a decrease in turbidity in the upper water column due to flocculation.



Line	Location	Date	n	(f)x	r^2
A.	Mare Island Strait	Sep-Oct 74	104	55.91-9.22 lnx	0.56
B.	Mare Island Strait	Mar 75	24	7.87-1.41 lnx	0.46
C.	Mare Island Strait	Oct 75	30	46.33-8.30 lnx	0.54
D.	Alameda NAS	Dec 74	38	120.59-15.58 lnx	0.26
E.	Alameda NAS	Jan 75	99	117.09-20.66 lnx	0.42
F.	Richmond Harbor	Nov 74	83	135.17-21.22 lnx	0.35
G.	Petaluma Channel	Sep 77	268	114.66-18.68 lnx	0.75

(1) f(x) = % transmission 10 cm light path.
(2) Samples greater than 500 mg/l suspended solids excluded.
(3) Samples 0% transmission entered at 0.01%.

Figure 7.

The turbidity associated with sand is limited to the percent silts and clays in the sediment and the type and size of equipment. The type and size of equipment relates to the disturbances and the absorption ratio. The absorption ratio is the ratio of sediment removed divided by sediment disturbed. The hopper dredge, working with sand, does introduce grading of sediments during dredging. Fines are separated through the overflow. The reduced fines level in the hopper does result in lower turbidity during disposal. This also is reflected in the contaminant levels. Contaminants in the disposed sediments are less than the _in situ_ project sediments. A large clamshell dredge will maintain a greater percentage of fines for transport. The turbidity during disposal, however, will be greater than from a hopper dredge because the fines are still present. The turbidity from a hydraulic cutterhead will depend on the ratio of absorbed sediments, which in turn relates to the suction's pickup capacity, swing speed, working face and type of sediment. The four elements are interdependent.

CONCLUSIONS

The type of sediment being dredged is a major parameter in evaluating both the disturbance generated by the operation and the efficiency of the operation. The disturbance and the efficiency are related. With silts and clays a less efficient operation means that more energy is being expended in adding and mixing water with the sediment. As the disturbance increases, the efficiency continues to decrease.

Many studies have been conducted during the past several years to define the disturbance during dredging. The great variation of results within and between studies probably could be correlated if the type of sediment, the shape and condition of the shoal, the type of equipment, the method of operation and relative time of sampling were entered into the analysis. The liquid limit and water content (presented as multiples of the liquid limit) of the sediments at any particular time and location of the operation are proposed as a common basis for evaluating levels of suspended solids in the water column.

Continuous "optimum" efficiency of a dredging operation can not be expected. Other constraints override the optimum conditions, such as the availability of a particular type and size of dredge, requirements to minimize interruptions of commerce, localized channel requirements and safety. Some of the factors can be controlled or developed to increase or maintain the efficiency of the operation. Training operating personnel to become familiar with the equipment in various types of sediments and shoal configurations is probably the most effective. Use of instrumentation for horizontal and vertical control will also increase efficiency. With hydraulic systems such as the hopper dredge and the cutterhead dredges, recently developed instrumentation should be utilized to indicate immediate pumping response to both increases and decreases in pumping densities. Clogging of pumps or breakage of sediment suction will be reduced and a more "optimum" density will be maintained. The design of the dredging operation should minimize feather shoals and cleanup requirements. This may mean an evaluation of neat line payment schedules in favor of some combination of neat line and pumpage quantities.

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BIOACCUMULATION OF TOXIC SUBSTANCES FROM CONTAMINATED SEDIMENTS BY FISH AND BENTHIC ORGANISMS

Robert M. Engler
Environmental Laboratory
Dredged Material Research Program
U.S. Army Engineer Waterways Experiment Station
Vicksburg, Mississippi 39180

BACKGROUND

The River and Harbor Act of 1970 (Public Law 91-611, Section 123) authorized the Corps of Engineers to initiate and conduct a comprehensive nationwide study of dredging and dredged material disposal operations. Of particular interest were environmental impacts, productive uses of dredged material, and new and/or improved dredging and disposal practices.

The U.S. Army Engineer Waterways Experiment Station (WES) was assigned responsibility for the research program; the program was designated as the Dredged Material Research Program (DMRP).

The planning and implementation of the DMRP were the responsibility of an interdisciplinary team established at WES as part of the Environmental Laboratory (EL). The thrust of the program involved four major research projects (1):

a) Environmental Impacts and Criteria Development Project (EICDP).

b) Habitat Development Project.

c) Disposal Operations Project.

d) Productive Uses Project.

This review report is primarily concerned with the findings from Task 1D, the Effects of Dredging and Disposal on Aquatic Organisms, and from Task 1A, the Aquatic Disposal Field Investigations (ADFI), of the DMRP and observations from two of the five open-water disposal sites. Findings from related work will be referenced in this review as appropriate.

The overall objective of the EICDP was to provide definitive information on the environmental impact of dredging and disposal operations and, where undesirable impacts were observed, to suggest means of eliminating or reducing such impacts. As such, this also included studies on water and sediment quality and the rate and extent of the recolonization of disposal sites by bottom organisms, impacts such as bioaccumulation of toxic substances on bottom animals, and responses of swimming and free-floating organisms to disposal.

Task 1D (2) included six research efforts that dealt with the direct and indirect effects of dredging and disposal on aquatic organisms. The aspects of dredging and disposal investigated for potential environmental effects were the physical disruption of the bottom environment, the generation of suspended sediments, and the contaminant load of the sediments being disturbed and redistributed.

The research reviewed is in the forefront of applied environmental science and is a beginning in defining the occurrence of environmental perturbations due to dredging and disposal. Most of the studies reviewed describe worst-case experimental conditions. Although somewhat limited in scope, experimental laboratory results showing lack of effects under these conditions support the conclusion of the field studies that indirect (long-term and sublethal) effects of dredging and disposal will be minimal.

Potential environmental effects of dredging and disposal are not yet completely understood due to the many variables involved. Dredging and disposal operations are carried out in many geographic locations with a wide range of aquatic environments. Waters may have different salinity regimes and different levels of natural turbidity. Disturbed areas may have different contaminant burdens in the water and sediments. Major variables are the presence of organisms and the species diversity that characterize the different dredging and disposal sites. Even different methods of dredging and disposal may affect the environmental impact of a given project.

The basic approach involved the selection of field sites on the basis of representativeness of different geographic regions (environments) and disposal operations. Appropriate strategies were then developed for the collection and analysis of biological, chemical, and physical samples. Samples were then taken during controlled disposal operations and compared to samples obtained under baseline conditions and from reference sites.

FIELD STUDY RESEARCH SITES

Lake Erie (Ashtabula, Ohio)

Seasonal aspects of spring and summer hopper dredge disposal of contaminated and uncontaminated sediments from Ashtabula Harbor were investigated at this site. In addition, the long-term impacts at a historic disposal site were evaluated. This was the only site located entirely in freshwater.

Duwamish Waterway (Puget Sound, Washington)

This site was chosen for investigations of the disposal of contaminated sediments in an estuary. Fine-grained sediments contaminated with polychlorinated biphenyls (PCBs), metals and petroleum hydrocarbons were mechanically dredged from the waterway, barged to an Elliott Bay (Puget Sound) site, and disposed of in 60 m of water.

RESEARCH RATIONALE

In general, a multidisciplinary team approach was used at each site to investigate the effects of dredged material disposal (3). The primary variables studied were physical, chemical, and biological parameters as follows:

Physical: Currents, waves, tides, meteorology, bottom profiles, sediment movement, sedimentology, geochemistry, and mineralogy.

Chemical: Water quality, sediment quality, toxicant release/removal, and nutrient release/removal.

Biological: Fish, shellfish, benthic macroinvertebrates, phytoplankton, zooplankton, contaminant bioaccmulation, and recolonization.

Not all variables were investigated at each site, however, this review deals only with the contaminant uptake potential for select toxic substances by aquatic organisms.

Three other task areas within the EICDP (1) were closely related to this work. These were: Movements of Dredged Material (Task 1B), Effects of Dredging and Disposal on Water Quality (Task 1C), and Effects of Dredging and Disposal on Aquatic Organisms (Task 1D). The research for these tasks was, for the most part, carried out in the laboratory under controlled conditions. As such, the results are useful for understanding known impacts and for predicting others that may occur. They cannot, however, be directly applied to field conditions without verification but can be considered as "worst case" evaluations. As such, they are useful in defining boundary conditions expected with aquatic discharge.

Results obtained in the field studies may be site-specific. Dredging and disposal will almost always cause some degree of environmental disruption. Disposal, for example, will usually cause the burial of organisms.

The apparent absence of an impact does not definitively demonstrate that one did not occur. Rather, it may reflect a deficiency in experimental design, inappropriate methods, or analytical error. This is a particular problem in the case of chronic or long-term impacts because these may not become evident for months or years after the causal event.

LABORATORY INVESTIGATIONS

Dredging and disposal are carried out in different locations throughout the United States and in the territorial sea. Very often disposal is carried out in an environment different from the dredging site. Dredging and disposal occur in waters ranging from fresh to estuarine and high salinity waters. Some of these waters are highly turbid, whereas others normally are quite clear. Different dredging areas may have different contaminant burdens in waters and sediments. Another major variable is the organisms present and the species diversity in different dredging and disposal areas. Some benthic substrates may host hundreds of species within areas of a few square meters while other substrates may host very few species. The presence of many different species at a given location has been classically interpreted as the sign of a healthy ecosystem. This is not always true, and it is important to also take into account the number of major different types of organisms. For example, in the Oakland Inner Harbor of San Francisco Bay, there is great animal diversity, but most of the different species are sludge worms, with a number of other species which are diminuitive, opportunistic, and adapted to pollutant stress. Conversely, potentially damaged species may be of commercial value, i.e., the east coast oyster beds. Local pollutant additions unrelated to dredging may exceed potential dredging and disposal effects or may act synergistically with dredging and disposal to produce deleterious or beneficial environmental effects. Even different methods of dredging and disposal may affect the environmental impact of any given project.

Previous literature on dredging and disposal has been fragmentary and site or resource specific. Reviews of literature reveal that too few basic data form a broad scale conceptual framework for the possible environmental impacts.

CHEMICAL IMPACTS

The potential availability and uptake of sediment-associated heavy metals by deposit-feeding benthic animals was the subject of DMRP Work Unit 1D06 (4). This laboratory study attempted to determine the biological availability of selected heavy metals associated with sediments and the potential for their uptake from dredged material into the tissues of representative bottom-dwelling species.

Sediment chemistry in relation to organisms impact was further studied in Work Unit 1D11 (5). This study carried out laboratory experiments on the transfer of oil and grease residues from oil-contaminated sediments into the tissues of some representative benthic species.

There has been concern that contaminants from municipal, industrial, and other sources which become entrapped in aquatic sediments may become biologically active when disturbed by dredging or disposal activities. Major materials in question have been numerous heavy metals, persistent pesticides such as DDT and its derivatives, PCBs and petroleum hydrocarbons. Dredging and disposal do not introduce new contaminants to the aquatic environment, but at worst simply redistribute the sediments which are the natural depository of contaminants introduced from other sources.

Toxic Substance Uptake

In Work Unit 1D06 (4), metal availability and accumulation studies were conducted using the clam *Rangia cuneata*, the grass shrimps *Palaemonetes pugio* and *P. kadiakensis*, and the worms *Neanthes arenaceodentata* and *Tubifex* sp. Test sediments, as shown in Table 1, were taken from Texas City and Corpus Christi, Texas, ship channels (15 and 30 $^{o}/_{oo}$ salinity, respectively) and the Ashtabula River in Ohio (freshwater). Metals routinely measured were iron, manganese, copper, cadmium, nickel, lead, zinc, chromium, and mercury.

TABLE 1. TOTAL CONCENTRATION OF METALS IN SEDIMENTS

	Total Concentration, mg/kg of Dry Weight		
Element	Texas City Channel	Corpus Christi Channel	Ashtabula River
Cu	48	120	37
Cr	188	82	175
Cd	2.4	21	4.8
Fe	14,500	12,000	25,000
Ni	48	17	52
Mn	570	257	356
Pb	41	316	42
Zn	161	4,055	315
Hg	0.6	18	1.1
V	136	--	222

Elutriate Test Results

	Texas City		Corpus Christi		Ashtabula	
Metal	Site water (μg/l)	Elutriate (μg/l)	Site water (μg/l)	Elutriate (μg/l)	Site water (μg/l)	Elutriate (μg/l)
Cu	20	9	9	3	8	6
Zn	44	28	325	1700	85	50
Mn	32	5800	22	890	3	550
Fe	44	52	10	20	15	650
Pb	1	1	2	6	1	1.5
Cr	7	6	<5	<5	<5	<5
Cd	<1	<1	<1	<1	<1	<1
Ni	85	75	11	9	21	20
Hg	0.05	0.10	<0.05	0.55	0.11	2.85

For most metals studied (Table 2) uptake by organisms was not evident. However, when uptake was shown to occur, the levels often varied from one sample period to another and were quantitatively marginal, usually being less than one order of magnitude greater than levels in the control organisms even after 1 month of exposure. It is invalid to compare metals levels in organisms to total sediment chemical concentrations since only a variable and small amount of the sediment-associated metal is biologically available. This is discussed in detail in the synthesis report on DMRP Task 1E (6). In addition to not knowing the amount of metal available for biological uptake, animals in undisturbed environments may naturally have high and fluctuating metal levels. Therefore, comparisons should be made between control and experimental organisms at the same point in time in order to evaluate bioaccumulation.

TABLE 2. SUMMARY OF BIOAVAILABILITY OF SEDIMENT-ADSORBED HEAVY METALS TO BENTHIC INVERTEBRATES DURING SHORT-TERM EXPOSURES

	Texas City Sediment			Corpus Christi Sediment			Ashtabula Sediment		
Metal	Rangia	Palaemonetes	Neanthes	Rangia	Palaemonetes	Neanthes	Rangia	Palaemonetes	Tubifex
Cd	0	-	0	0	0	0+	0	0	0
Cr	0	+	+	0	0	0	0	0	0
Cu	0	0	+	0	-	0+	0	0	0
Fe	0	+	0	0	+	0+	+	+	+
Hg							0	0	
Mn	+	+	0	0	-	0	0	0	0
Ni	0	-	0	-	-	0	0	-	0
Pb	0	0	+	0	+	+	+	0	+
V							+	+	
Zn	0	-	0	0	+	0+	0	-	-

+ = Statistically significant effect of exposure, and/or the interaction of exposure with salinity, time, or both on the concentrations of the heavy metal in the animal's tissues.

- = Statistically significant inverse effect of exposure, and/or the interaction of exposure with salinity, time, or both on the concentrations of the heavy metals in the animal's tissues (control animals had significantly higher metal concentrations than did sediment-exposed animals).

0 = Exposure to sediment did not contribute significantly to the concentration of the heavy metal in the animal's tissues.

0+ = Replicate experiments with Neanthes exposed to Corpus Christi sediment at 30 $^{\circ}/_{\circ\circ}$S gave opposite results.

Statistical analyses of Hg and V were performed only on tissues of Rangia and Palaemonetes exposed to Ashtabula sediment in freshwater. Hg analyses for other tests were at or below detection limits with few exceptions and V was only studied for Ashtabula sediments.

Of a total of 168 animal-sediment-salinity combinations evaluated in tests carried out by Neff, Foster, and Slowey (4), only 22 percent showed significant accumulation due to sediment exposure. The largest uptake was of iron, a metal generally known for its low degree of toxicity in biological systems. Significant accumulations of lead were seen in a number of short-term exposures, although these could not be duplicated in long-term exposures. Relatively high uptake of lead occurred only in the polychaete Neanthes and was interpreted to be potentially ecologically significant for this species. Their literature search showed that heavy metals in solution vary over several orders of magnitude in availability to benthic invertebrates. Although accumulation of heavy metals by organisms from the water has been documented, the literature shows no such clear evidence for accumulation of metals from the sediments.

Neff, Foster and Slowey (4) also investigated the depuration of heavy metals after the organisms were removed from the test sediments. In those 37 cases where there was uptake after 8 days exposure, depuration during 2 or 8 days in clean water was seen in 7 instances, with the other 30 cases showing no decrease in metal concentration in the tissues.

In a field investigation of the San Francisco Bay system, Anderlini et al. (7) looked at 9 heavy metals (silver, arsenic, cadmium, copper, mercury, nickel, lead, selenium, and zinc) and 5 invertebrates (Ampelisca milleri, Macoma balthica, Neanthes succinea, Mytilus edulis, and Ischadium demissum). Metals concentration in sediments and organisms fluctuated within and outside the dredged zone during the period of the study. Changes in the mean metal concentrations in sediments and all invertebrates during the study period were relatively small, considerably less than one order of magnitude. Mean metal concentrations in sediments and benthic invertebrates changed by less than a factor of 2, and changes in metal levels in M. edulis were no greater than a factor of 3. These changes could not be directly attributed to dredging activities. Metal concentrations were similar in M. edulis which were transplanted from clean water to stations within and outside the dredged zone. Mussels transplanted to contaminated Bay stations appeared to accumulate copper, nickel, and zinc over controls kept in clean water coastal stations but to a lesser extent than native mussels. Desorption of metal species by mussels 27 days after being transferred from Bay or ocean stations occurred in the following order of decreasing depuration: zinc > mercury > copper > lead > nickel > cadmium > arsenic. Selenium was not depurated from mussel tissue in 27 days.

The accumulation potential of a metal may be affected by several factors such as duration of exposure, salinity, water hardness, exposure concentration, temperature, and the particular organism under study. The relative importance of these factors varies from metal to metal. Data of Neff, Foster, and Slowey (4) on salinity effects are inconclusive, but there was a trend toward increased uptake at lower salinities. Anderlini et al.'s (7) 9-day laboratory study exposed M. balthica to the chloride salts of various metals in the water column. These data support field observations in which M. balthica showed the highest metal concentrations following dredging periods where heavy rains had resulted in a marked decrease in salinity.

The Neff, Foster, and Slowey (4) study indicated that the chemical form of metals had important effects on their bioavailability. Elevated concentrations of heavy metals in tissues of benthic invertebrates were not always indicative of high levels of metals in the ambient medium or associated sediments. Although a few instances of uptake were seen to be of possible ecological significance, diversity of results among species, different metals, types of exposure, and salinity regimes strongly argued that bulk analysis of sediments for metal content could not be used as a reliable index of metal availability and potential ecological impact of dredged material.

Neff, Foster, and Slowey (4) performed sequential and nonsequential chemical extractions on the sediments to evaluate the potential mobility of metals in different chemical forms. They also determined the total metal concentration in the sediment. For some species a correlation did exist and for others a correlation did not exist between any chemical or physical form studied and bioaccumulation of the metal. These authors state: "At present, it does not appear that a simple extraction scheme can be developed that might indicate availability of sediment sorbed metals by benthic organisms. Additional data, based upon a large number of different sediment types, may indicate, however, forms most likely to be accumulated by benthic organisms."

For some metals there appears to be good correlation between metal concentration in the sediment and in the associated infaunal and epifaunal macrobiota (4). For other metals no such correlation exists. These correlations often vary with sediment type. The correlation, when it occurs, may be due to direct or indirect transfer of metals from sediment to biota or it may represent the presence of a common source of metals to both the sediment and biota. Anderlini et al. (7) concluded that if changes in metals in the water occurred as a result of dredging activities, the changes were either less than small natural fluctuations or were of short duration.

Both Neff, Foster, and Slowey (4) (short-term laboratory studies and literature review) and Anderlini et al. (7) (longer term field work and back-up laboratory experiments) have found the same heavy metal phenomenon. The accumulation and release of certain heavy metals seems to vary with the metal, with the species, between sampling times, between sampling sites (dredged and not dredged), and within controls. These variable results have not been directly correlated with dredging operations or sediment loading.

A recent field study supporting the laboratory results of Neff, Foster, and Slowey (4) has been carried out by Simms and Presley (8). These authors concluded that mollusks, crustaceans, and bony fishes from dredged areas of San Antonio Bay were lower in almost every heavy metal than were organisms from other areas where dredging was minimal. Mollusks were observed to concentrate metals more than any other organisms studied, but the levels observed were much lower than those thought to be lethal or toxic. Except for a few large fish, metal concentrations did not correlate significantly with size or growth stage. Vigorous shell dredging in the Bay for 50 years apparently did not cause increases of heavy metals in the tissues of local biota.

Studies in DMRP Work Unit 1D09 (9) used harbor sediments chosen for physical similarity to bentonite, in order to assay for impacts due to chemical properties of the sediments in suspension. Measurements were carried out using sediments from relatively uncontaminated reaches of San Francisco Bay and compared with measurements on more highly contaminated Bay sediments. Organism responses did not differ greatly between pure mineral suspensions (10) and uncontaminated natural sediments. In many cases, lethal effects were more marked with the contaminated sediments. Sediment characteristics are presented in Table 3. The most sensitive species tested, striped bass, Morone saxatilis, survived only a few hours at levels of 0.5 g/l of contaminated sediments, a condition probably representing a worst-case of turbidity generation associated with a dredging operation. Such conditions are very unlikely to occur in the field, where motile organisms may escape turbidity maxima, and where water currents disperse sediments as they settle out of the water column.

Chemical analyses of several species for heavy metals, pesticides, and polychlorinated biphenyls (PCBs) as presented in Tables 4-7, indicated uptake of several contaminants, but none were accumulated to levels which appeared to be sufficient to influence the survival of the exposed organisms (9). Difficulties in interpreting such chemical data argued for developing assays which evaluate total toxicity of a sediment regardless of specific toxicants.

Oil and Grease Uptake

This term is used collectively in describing all components of sediments of natural and contaminant origin which are primarily fat soluble. The literature review contained in DMRP Work Unit 1D11 (11) demonstrated a broad variety of possible oil and grease components in sediment, the recovery of which was dependent on the type of solvent and methodology used to extract these residues. Trace contaminants, such as the PCBs and chlorinated hydrocarbons (DDT and derivatives), often occur in the oil and grease. Large amounts of contaminant oil and grease find their way into the sediments of the Nation's waterways either by spillage or as chronic inputs in municipal and industrial effluents, particularly near urban areas with major waste outfalls. The literature suggested long-term retention of oil and grease residues in sediments with minor biodegradation occurring. Where oily residues of known toxicity became associated with sediments, these sediments retained toxic properties over periods of years affecting local biota. Spilled oils are known to readily become adsorbed to naturally occurring suspended particulates, and oily residues in municipal and industrial effluents are commonly found adsorbed to particles. These particulates are deposited in benthic sediments and are subject to resuspension during disposal.

Using the elutriate test DiSalvo *et al*. (11) showed some release into the water of soluble hydrocarbon residues from sediments known to contain 2000 to 6000 ppm total hydrocarbons. Hydrocarbon concentrations in the elutriate (100 to 400 ppb) were from 11 to 400 times higher than background, yet were well below acceptable effluent discharge standards. The amount of oil released during the elutriate test is less than 0.01 percent of the sediment-associated hydrocarbons under worst-case conditions.

A test scheme was employed in which estuarine crabs (Hemigrapsus oregonensis), mussels (Mytilus edulis), and snails (Acanthina spirata), and

TABLE 3. CHEMICAL CHARACTERIZATION OF CONTAMINATED SEDIMENT USED IN THE ESTUARINE AND MARINE TESTS

	Sediment Fraction			
Parameter	Total mg/kg	Exchangeable mg/kg	Interstitial Water - mg/l	Elutriate mg/l
% water	48.3	--	--	--
pH	7.8	--	--	--
Eh	-414	--	--	--
Total Sulfides	6148	--	--	--
Total Phosphates	878	--	--	--
Ortho-P	--	--	2.3	1.4
TKN	0.15%	--	--	--
HN_4^+-N	--	88.8	--	(3.49)
As	128	0.50	0.12	0.14
Cd	2.3	1.09	0.16	(0.14)
Cu	158	1.6	0.10	0.06
Fe	3.62%	--	2.5	0.18
Mn	333	114	5.1	0.49
Hg	1.47	0.55	0.15	(0.16)
Ni	104	62.4	6.4	9.6
Se	1.49	0.62	0.48	0.46
Zn	381	4.0	0.12	0.04
Total PCBs	1.30	--	--	--
Total DDT	0.750	--	--	--

TABLE 4. CONTAMINANT CONCENTRATIONS IN THE TISSUE OF THE MUSSEL *Mytilus edulis* IN THE CHEMICAL UPTAKE STUDY IN THE CONTAMINATED SEDIMENT PHASE OF THE ESTUARINE TEST

SS g/l	Exposure days	Metals (μg/g wet)								
		As	Cd	Cu	Fe	Pb	Mn	Hg	Ni	Zn
At collection		0.09	0.96	2.05	71	0.19	2.71	0.15	0.68	38.4
0	3	0.15	0.09	5.98	176	0.15	7.74	0.12	1.24	45.6
0	10	0.15	0.32	1.74	131	0.19	3.29	0.05	0.94	25.0
0	15	0.28	0.26	1.16	100	0.23	1.16	0.02	1.74	20.2
0	21	0.6	0.44	1.14	95	0.18	3.09	0.06	2.60	39.1
0	P	0.06	0.28	0.85	49	0.15	3.49	0.02	3.15	27.5
3.6	P	0.16	2.15	5.22	154	1.68	2.99	*	2.72	100.0
12.1	P	0.10	1.16	2.43	55	0.34	2.92	*	1.46	55.9
15.9	P	0.08	1.07	2.83	58	0.18	1.95	0.11	1.17	50.0

P = Animals were exposed to indicated suspended solids concentration for 21 days then placed in clear water 5 days to purge the sediment from the digestive tract and body surfaces before analyses.

* = Insufficient sample for analysis.

TABLE 5. WHOLE BODY CONCENTRATIONS OF SELECTED METALS IN CHEMICAL UPTAKE STUDY WITH CONTAMINATED SEDIMENT IN THE MARINE TEST

Species	Suspended Solids Exposure	Metals (μg/g wet)								
		As	Cd	Cu	Fe	Pb	Mn	Hg	Ni	Zn
Mytilus edulis	Control	0.12	0.34	2.48	154.0	2.14	2.53	0.63	0.04	25.7
	Purged	0.20	0.10	7.66	215.0	3.89	2.48	0.02	0.05	33.6
Mytilus californianus	Control	0.17	0.51	3.52	162.0	0.97	1.62	0.21	0.08	22.7
	Purged	0.12	0.45	9.23	88.8	2.28	0.95	0.03	0.004	25.0
Crangon nigromaculata	Control	0.03	0.06	4.95	38.8	0.55	0.87	0.13	0.01	7.2
	Purged	0.08	0.23	9.08	25.6	0.27	8.20	0.14	0.52	22.7
Cancer magister	Control	0.03	0.10	16.8	55.9	0.31	1.00	*	0.19	13.3
	Purged	0.03	0.13	13.3	51.4	0.38	8.06	*	0.37	16.8

* = Insufficient sample for analysis.

TABLE 6. CONTAMINANT CONCENTRATIONS IN THE TISSUE OF THE MUSSEL *Mytilus edulis* IN THE CHEMICAL UPTAKE STUDY IN THE CONTAMINATED SEDIMENT PHASE OF THE ESTUARINE TEST

SS g/l	Exposure days	Chlorinated Hydrocarbons (µg/g wet)							
						Aroclor			
		DDE	DDD	DDT	Total DDT	1241	1254	1260	Total PCBs
At collection		0.68	0.32	-	1.00	0.02	0.05	0.01	0.08
0	3	1.18	0.38	-	1.56	0.02	0.06	0.01	0.09
0	10	0.52	0.84	-	1.36	0.02	0.03	0.01	0.06
0	15	0.50	0.46	-	0.96	0.02	0.06	0.02	0.10
0	21	0.18	0.85	-	1.03	0.04	0.02	0.01	0.07
0	P	0.55	0.40	-	0.95	0.06	0.02	0.01	0.09
3.6	P	0.75	0.83	-	1.58	0.07	0.07	0.02	0.16
12.1	P	0.49	1.10	-	1.59	0.05	0.03	0.02	0.10
15.9	P	0.47	0.68	-	1.15	0.05	0.02	0.01	0.08

P = Animals were exposed to indicated suspended solids concentration for 21 days then placed in clear water 5 days to purge the sediment from the digestive tract and body surfaces before analyses.

- = See note Table 7.

TABLE 7. WHOLE BODY CONCENTRATIONS OF SELECTED CHLORINATED HYDROCARBONS IN CHEMICAL UPTAKE STUDY WITH CONTAMINATED SEDIMENT IN THE MARINE TEST

Species	Suspended Solids Exposure	Chlorinated Hydrocarbons (µg/g)			
		DDE	DDD	DDT	DDT
Mytilus edulis	Control	1.71	-	0.69	2.40
	Purged	2.02	-	-	2.02
Mytilus californianus	Control	1.66	-	-	1.66
	Purged	2.57	-	-	2.57
Crangon nigromaculata	Control	1.54	-	-	1.54
	Purged	1.39	-	-	1.39
Cancer magister	Control	0.14	-	-	0.14
	Purged	0.14	-	-	0.14

- = Below detection limits; DDD < 0.006 ng/g; DDT < 0.008 ng/g. Also below detection limits were aldrin, dieldrin, heptachlor <0.004 ng/g; chlordane < 0.008 ng/g; endrin, PCBs < 0.1 ng/g.

the freshwater clam, Corbicula sp., were exposed to contaminated sediments in order to determine magnitudes of uptake of hydrocarbons which were included in sedimentary oil and grease burdens. Sediment characteristics are shown in Table 8.

There was no overt mortality of test organisms that was directly attributable to exposure to contaminated sediments. Experimental evidence as presented in Table 9 suggested slight uptake of hydrocarbons by saltwater test organisms incubated in the presence of Duwamish River sediments which contained almost 500 ppm total hydrocarbons (11). Freshwater clams exposed for 30 days to Duwamish River sediments showed no well-defined uptake of hydrocarbons (11). As presented in Table 10, mussels and crabs exposed for 4 days to New York Harbor sediments containing 2000 ppm total hydrocarbons showed average uptakes above background of about 50 to 70 ppm (2.5 and 3.5 percent, respectively, of the sedimentary hydrocarbon concentration).

These results (11) indicated that selected estuarine and freshwater organisms can be exposed to dredged material that is contained with thousands of parts per million oil and grease and experience minor mortality for periods up to 30 days. Uptake of hydrocarbons from the heavily contaminated sediments appears minor when compared to the hydrocarbon content of the test sediments and when compared to results describing exposure of uncontaminated organisms under field conditions where total hydrocarbon uptake ranged to several hundred parts per million (12).

In Work Unit 1D07 (13), attempts were made to trace pathways of uptake of sediment-associated DDT into the tissues of estuarine deposit-feeding benthic infauna. The data obtained suggested the possibility of uptake of DDT under model laboratory conditions which may or may not be operative under field conditions. Fulk, Gruber, and Wullschleger (14) have reviewed the literature on pesticides and PCBs in sediments. Algae, suspended solids, bottom sediments, and water contain various chlorinated hydrocarbons. The studies conducted on the adsorption and desorption of chlorinated hydrocarbons on solids have generally indicated that the materials are much more readily sorbed than desorbed. These workers analyzed the sediments from five locations for aldrin, dieldrin, endrin, lindane, 2,4-D esters, DDT analogs, toxaphene, and PCBs. PCBs, dieldrin, and the DDT analogs were the most prevalent. The desorption of the latter materials was studied. No release of DDT residues was observed. Some dieldrin release was observed in the parts per trillion range. On the basis of these laboratory studies, it appears that release of these water-insoluble pesticides will not occur to an appreciable extent during disposal. In another study, Anderlini et al. (15) monitored release from sediments and uptake by organisms of PCBs and compounds of the DDT group during a disposal operation in San Francisco Bay. Some uptake of p,p′-DDE was observed but the levels of the other chlorinated hydrocarbons remained constant in Mytilus edulis.

TABLE 8. CHARACTERISTICS OF SEDIMENTS OBTAINED FOR STUDY

Sample	Dry Solids % sediment	Volatile Residue % dry solids	Oil and Grease % dry solids	Hydrocarbons % oil and grease
Duwamish River (Slip 1)	49.0	7.1	0.12	52
Duwamish River	36.6	6.4	0.23	37
Oakland Middle Harbor	51.6	7.8	0.07	62
New York Harbor (Bay Ridge)	30.1	10.2	0.88	68
New York Harbor (Perth Amboy)	34.4	13.0	0.45	63

(μg/g dry weight)

Sample	A Oil and Grease	B Total Hydrocarbons	C Hydrocarbons Desulfurized	(B-C) Sulfur	(A-B) Polar Materials
Duwamish River (Slip 1)	1224	637	338	299	587
Duwamish River	2301	854	413	441	1447
Oakland Middle Harbor	748	462	275	187	286
Oakland Middle Harbor (aged)	1050	710	93	617	340
New York Harbor (Bay Ridge)	8755	5930	4145	1785	2825
New York Harbor (Perth Amboy)	4525	2850	1580	1270	1675

TABLE 9

HYDROCARBON ANALYSES BY TLC OF CRABS (*Hemigrapsus oregonensis*) EXPOSED TO DUWAMISH RIVER DREDGED MATERIAL

Tank Series*	Time Days	Hydrocarbon Content µg/g dry tissue (ppm) Alkanes	Arenes	Total
	0	14	2	16
SED	17	22	7	29
SED, Ref	17	8	3	11
SCR	17	21	14	35
SCR, Ref	17	19	10	29
STR	17	25	6	30
STR, Ref	17	10	5	15

HYDROCARBON ANALYSES BY TLC OF MUSSELS (*Mytilus edulis*) EXPOSED TO DUWAMISH RIVER DREDGED MATERIAL

Tank Series*	Time days	Hydrocarbon Content µg/g dry tissue (ppm) Alkanes	Arenes	Total
	0	143	35	178
SED	30	35	58	93
SED, Ref	30	48	47	95
SCR	30	84	54	138
SCR, Ref	30	35	<6.5	<41.7
STR	30	110	52	162
STR, Ref	30	63	15	78

HYDROCARBON ANALYSES BY TLC OF SNAILS (*Acanthina spirata*) EXPOSED TO DUWAMISH RIVER DREDGED MATERIAL

Tank Series*	Time days	Hydrocarbon Content µg/g dry tissue (ppm) Alkanes	Arenes	Total
	0	<6	<4	<9
SED	30	14	7	21
SED, Ref	30	4	4	8
SCR	30		CONTAMINATED	
SCR, Ref	30	6	2	8
STR	30	28	37	65
STR, Ref	30		CONTAMINATED	

* = SED - Organisms in sediment.
SCR - Organisms on screen 5 cm above sediment.
STR - Organisms on screen 30 cm above stirred sediment.
Ref - Reference sediment.

TABLE 10

HYDROCARBON ANALYSES BY TLC OF CRABS (Hemigrapsus oregonensis) AND MUSSELS (Mytilus edulis) EXPOSED TO PERTH AMBOY (NJ) DREDGED MATERIAL

Tank Series*	Time days	Hydrocarbon Content µg/g dry tissue (ppm) Alkanes	Arenes	Total
Crabs				
	0	19	6	25
	0	21	7	28
STR #1	4	47	21	68
	4	59	34	93
STR #2	4	66	42	108
	4	85	35	120
Mussels				
	0	17	46	63
STR #1	4	22	65	87
	4	28	76	104
STR #2	4	18	86	104
	4	20	143	163

HYDROCARBON ANALYSES BY TLC OF CRABS (Hemigrapsus oregonensis) AND MUSSELS (Mytilus edulis) EXPOSED TO BAY RIDGE (NY) DREDGED MATERIAL

Tank Series*	Time days	Hydrocarbon Content µg/g dry tissue (ppm) Alkanes	Arenes	Total
Crabs				
	0	62	11	73
SED	1	58	22	80
SCR	1	24	5.2	30
STR	1	70	22	92
Mussels				
	0	25	22	46
SED	27	72	121	193
SCR	27	23	32	55
STR	27	108	180	288

* = SED - Organisms in sediment.
SCR - Organisms on screen 5 cm above sediment.
STR - Organisms on screen 30 cm above stirred sediment.

FIELD INVESTIGATIONS

FRESHWATER SITE

The Ashtabula, Ohio, ADFI site is located in Lake Erie (16) just north of the entrance to Ashtabula Harbor (Figure 1). The movement of surface water in the lake is counterclockwise although reversals do occur with northeast winds. A compensating current is found in the deeper waters of the lake during thermal stratification (June-October). Because of the configuration of the lake, any contaminants which are released along the south shore tend to move eastward along the shore. Oxygen depletion occurs in the deeper water during the summer.

Sediment in the disposal area primarily originates from material transported by the longshore current and, to a lesser extent, from the Ashtabula River which enters Lake Erie through Ashtabula Harbor. The sediment consists of about equal parts of sand and silt with a small amount ($\leq$10 percent) of clay. There is apparently little variation in grain size with depth.

Although there have been severe water quality problems in the lake, striking improvements have been noted in recent years. At the Ashtabula ADFI site, water quality variables tended to be quite uniform throughout the water column except during stratification. The expected differences resulting from stratification were observed; during periods of upwelling, deeper (hypolimnetic) water was often found quite near the surface.

A variety of invertebrates and fish inhabit the area. The former includes mollusks, worms, insect larvae, and crustaceans. These form a food supply for the 40-odd species of fish which were observed. Yellow perch were the most abundant species, with alewife, gizzard shad, and white sucker being quite common. Moderate to abundant populations of both zooplankton and phytoplankton occur throughout the lake.

The Ashtabula area is heavily industrialized and is a major port facility. There are a number of industrial and agricultural sources of contaminants in the immediate vicinity and there are two fossil-fuel generating stations east of Ashtabula. These discharge almost 2300 x 10^6 l/day of cooling and waste water into Lake Erie.

The Ashtabula ADFI spanned 3 years (1975-1977). The study consisted of four phases: a pilot survey (1975), predisposal sampling (1975), disposal operations (1975-1976), and postdisposal sampling (1975-1976). Samples for various parameters were obtained from a disposal area and an adjacent reference area.

The physical variables measured included currents, temperature, light transmission, meteorology, waves, bathymetry and sub-bottom profiles, grain size, sedimentology, and hydrology.

Both pumped samples and grab samples were obtained for analyses of conductivity, pH, turbidity, dissolved oxygen, nutrients, alkalinity, metals, organic carbon, silicates, and sulfate in the water column. Bulk sediment and

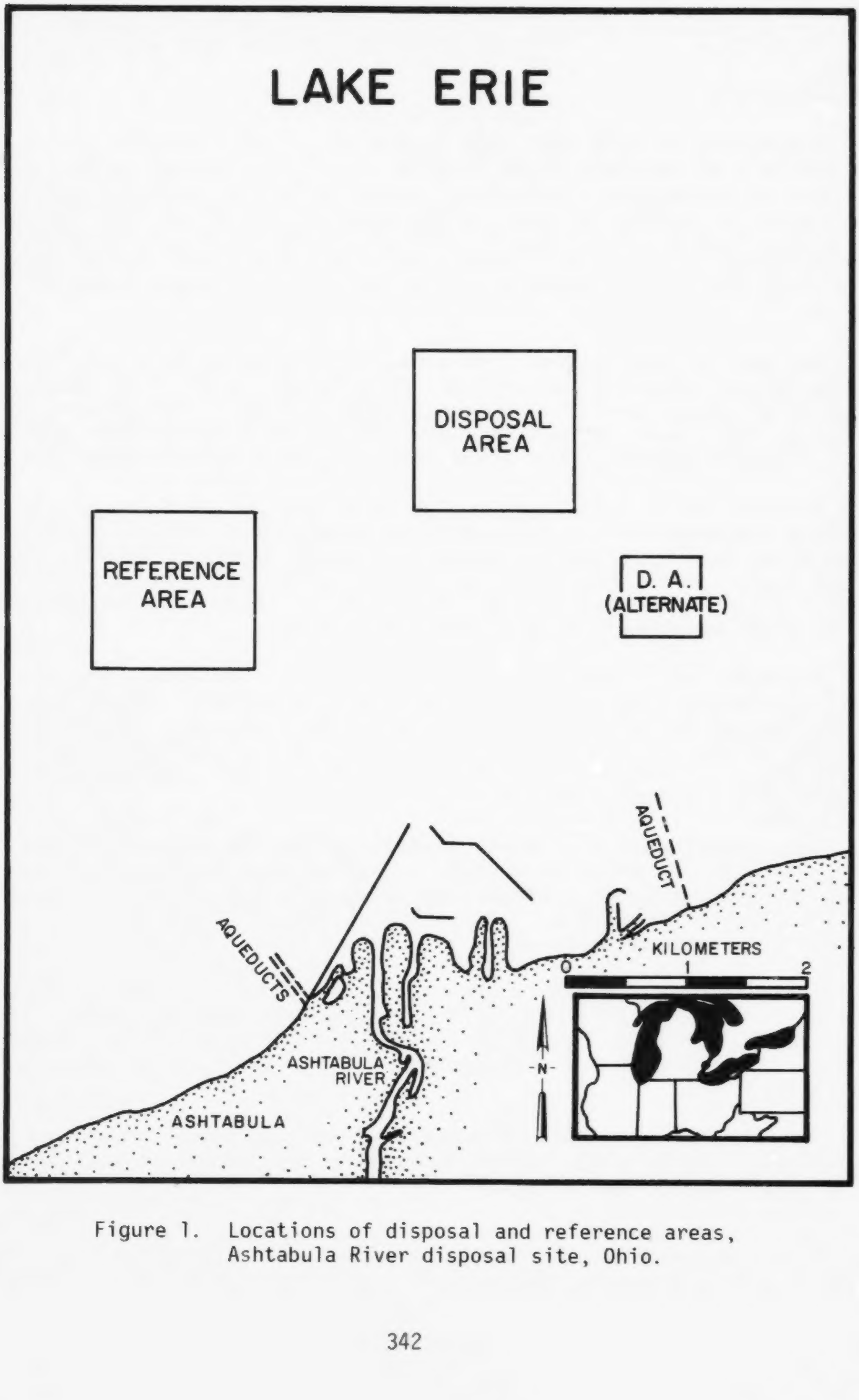
LAKE ERIE
DISPOSAL AREA
REFERENCE AREA
D. A. (ALTERNATE)
AQUEDUCT
AQUEDUCTS
KILOMETERS
0
1
2
N
ASHTABULA RIVER
ASHTABULA

Figure 1. Locations of disposal and reference areas, Ashtabula River disposal site, Ohio.

sediment interstitial water were analyzed for nutrients, metals and organic carbon; bulk sediments were further examined for pH, Eh, percent water, and cation exchange capacity. In addition, in-place sediment oxygen demand measurements were carried out.

Phytoplankton and zooplankton samples were evaluated in terms of species present and abundance; primary productivity was estimated by pigment analysis and carbon-14 uptake. Elutriate from dredged samples was added to phytoplankton samples to determine if inhibitory or stimulatory effects were present. Bottom grabs were obtained for investigation of macro- and meiobenthic organisms. As with plankton, these samples were evaluated to determine the numbers and kinds of organisms present.

A variety of fishery studies were carried out. These included sampling with gill nets and otter trawls, tows for fish larvae, age determination, and examination of stomach contents. Both fish and invertebrates were analyzed for heavy metals.

Findings

Spring and summer disposal by hopper dredge resulted in the formation of mounds of dredged material in the disposal area. These mounds (16) were 30 to 50 cm high, and, rather than a single mound being present, there were numerous small mounds. Disposal also created a small (< 2°C) transient increase of temperature in the water column; during thermal stratification, disposal did not alter the thermal structure. There was little change in grain size after disposal and those few changes observed had disappeared within 3 months. Erosion of the mounds occurred as a result of fall and winter storms, and there was a new transport of material to the northwest and southeast.

Almost all of the chemical variables measured in the water column were affected by disposal (16). Effects were not great, however, and an essentially complete return to ambient predisposal conditions was noted within a few minutes to several weeks. The overall impact of disposal is not clear as some constituents increased, presumably through release, while others decreased. The latter phenomenon probably resulted from sorption into settling dredged material.

There were changes in interstitial water (of sediment) chemistry after disposal (16). A return of predisposal conditions took from 30 to 90 days. It should be kept in mind that the sediments were eroding and being compacted and/or reworked after disposal. This process in itself could bring about various changes in interstitial water chemistry.

The greatest chemical effect of disposal appears to have been observed in the sediment. Following disposal, nutrients increased in the sediment, but metals (except mercury) decreased. This effect is not surprising as it reflects the relative concentrations of nutrients in lake sediments and harbor sediments.

Overall interpretation of the results of sediment chemistry are difficult because of the behavior of dredged material when released and of the natural lake sediments. Rather than there being an overlay of dredged material upon natural lake bottom, the physical impact of the dredged material striking the bottom resulted in bottom currents. These currents pushed lake bottom to the periphery of the study area and on top of previously deposited dredged material. Hence, alternating series of dredged material and natural bottom resulted, with subsequent compaction and reworking serving to further obscure differences between the two sediment types.

Disposal operations at Ashtabula had essentially no measurable impact upon planktonic organisms (16). Benthic organisms were impacted in several ways. There was no change in the number of species present in the disposal area following disposal, but there were a number of changes in species composition, with new species transported from the harbor replacing those which had been eliminated. In addition, there was a large increase in the number of organisms in the disposal area. Many of the changes did appear to be initially confined to the immediate area of disposal. As erosion spread the dredged material over a larger area, faunal changes in the expanded area were observed. Of interest was the finding that gross animal groups (such as the family level of identification) were not sufficient to determine impacts; rather, an examination at the species level was required.

Adults and young of pelagic fish did not appear to be impacted by disposal. However, bottom-dwelling fish showed a negative response to disposal and migrated from the area. Within an hour after disposal these fish had migrated back into the disposal area. Overall, the effects of disposal upon fish were of small magnitude and only persisted for a short period of time.

Heavy metals in fish and invertebrates, as presented in Tables 11 and 12, showed little change as a result of disposal. The relative concentration of metals in fish were the same as those observed in the sediment, whereas a decrease was noted in some of the invertebrates. Hence, bioaccumulation did not occur.

ESTUARINE SITE

The Duwamish River enters Elliott Bay, a part of Puget Sound (Figure 2). The entire river is tidal with horizontal and vertical variations in salinity. These depend upon tidal stage and river discharge. Low (<3-mg/l) dissolved oxygen concentrations occur near the bottom of the river. Although quite important as a waterway, the Duwamish is also a major migration route for salmon and trout.

Elliott Bay is a rather typical estuarine system with a surface layer of low salinity water being present over a deeper layer of more saline water. During the summer, density stratification is present but in the winter colder freshwater from the waterway entrains and mixes with warmer saline water. Hence, there is usually no stratification in the winter. Because it is an estuary, water column chemical constituents tend to be rather variable. The waterway has created an underwater delta along the south side of the bay. The deltaic sediments consist mainly of silty sand mixed with wood and other organic debris. The dominant demersal fish in the bay during the winter are assorted soles, and the dominant benthic invertebrate is the pink shrimp. Worms and various mollusks are also important components of the bottom fauna.

TABLE 11. OVERALL STATISTICS FOR WEIGHT AND METAL CONCENTRATIONS IN FISH GROUPS

		µg/g dry				
	1975	Reference Area Pre-75	Disposal Area Pre-75	Reference Area Post-75	Disposal Area Post-75	1 mi N CEI Area Post-75
Wt (g)	Mean	165.7	80.3	55.0	153.8	270.0
Hg	Mean	0.51	0.18	0.25	0.38	0.31
Fe	Mean	171	110	46	197	181
Pb	Mean	9.3	2.5	1.7	5.3	5.5
Cd	Mean	5.9	0.7	0.6	1.7	0.9
Cr	Mean	0.6	0.3	1.1	1.1	0.8
Ni	Mean	4.2	1.6	2.5	3.8	3.8
Cu	Mean	3.6	4.0	3.2	5.8	3.5
Mn	Mean	15.1	11.6	10.2	17.8	10.5

TABLE 12. COMPARISON OF METAL CONCENTRATIONS ON OLIGCHAETA BY AREA (ALL VALUES µg/g WET).

Metal	Reference Site All Cruises	Disposal Sites May (predisposal)	River Sites May (predredging)	Disposal Sites June (5-days post)	Disposal Sites July (30-days post)
Cd	0.72	0.19	0.03	0.13	0.13
Cu	11.0	11.6	3.0	3.7	2.7
Mn	18.9	11.1	7.9	7.5	9.1
Zn	152	54.6	21.0	57.2	27.4
Fe	1385	952	450	696	587

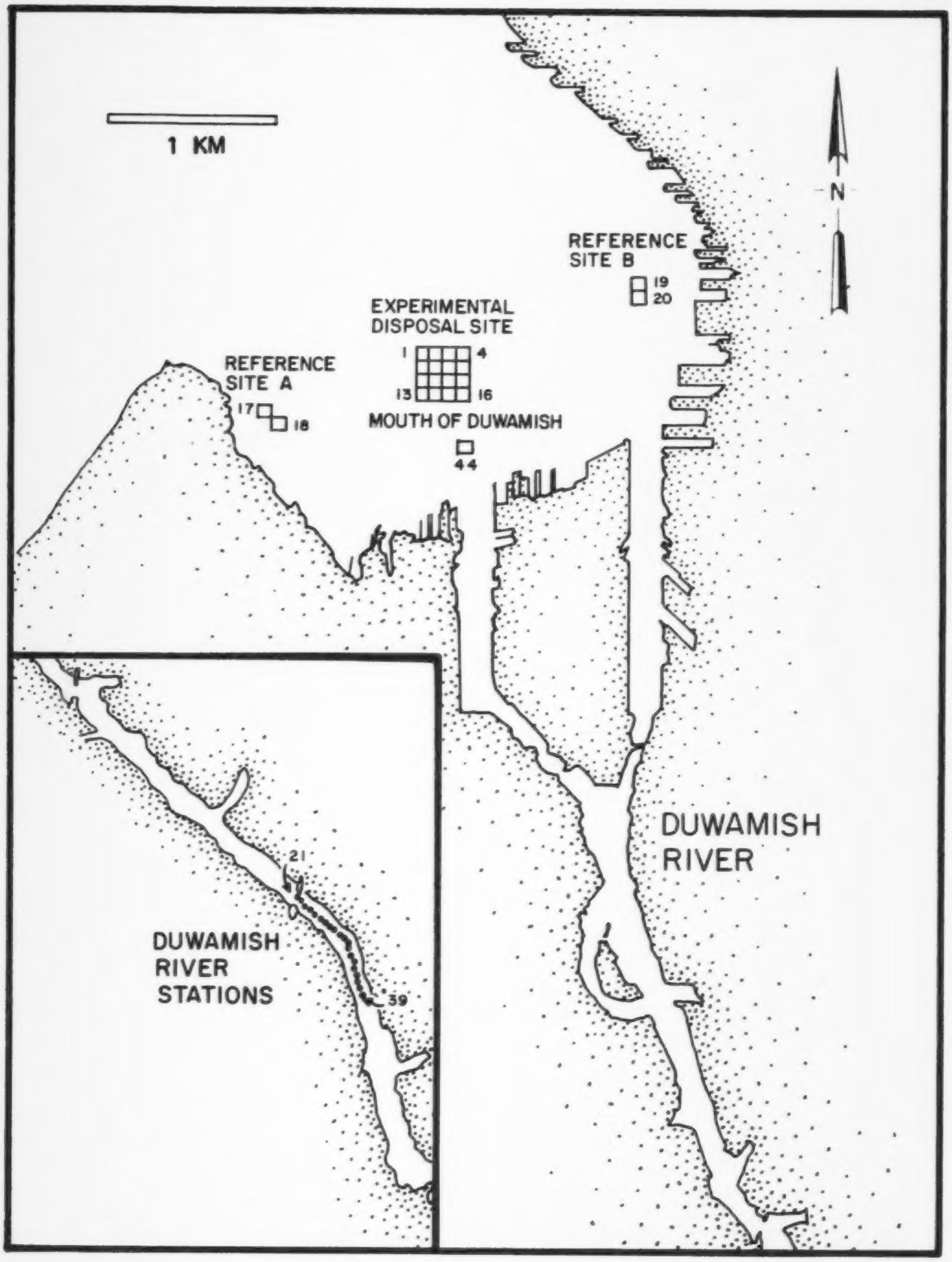
1 KM
N
REFERENCE SITE B
19
20
EXPERIMENTAL DISPOSAL SITE
1
4
13
16
REFERENCE SITE A
17
18
MOUTH OF DUWAMISH
44
DUWAMISH RIVER
21
DUWAMISH RIVER STATIONS
39

Figure 2. Locations of Duwamish Waterway disposal and reference sites, Puget Sound, Washington.

Dredging has ordinarily been done in the waterway with an hydraulic pipeline dredge and upland disposal has been used. However, the increasing cost of upland disposal required a shift to the use of mechanical dredging and open-water disposal with barges. In 1974, there was a spill of almost 1000 l of PCBs at Slip 1, in the maintenance dredging area of the river. The highly contaminated sediments were hydraulically dredged and placed in an impervious containment area (17) while the remaining, less contaminated material was removed by a clamshell dredge, placed in barges, and transported to the experimental disposal site. The dredging and disposal of the highly contaminated sediments were carefully monitored by the Environmental Protection Agency (EPA). The EPA found that there was a minimal release of metals, nutrients, and hydrocarbons (17).

The ADFI was divided into four phases: a pilot survey and predisposal, disposal monitoring, and postdisposal studies. During the pilot survey, an experimental disposal site was chosen for disposal, and two reference sites (to the east and west) were selected to provide comparative data (Figure 2). The studies were initiated in 1975 and completed in 1976.

Physical investigations conducted for the various phases included grain size analyses and measurements of currents, waves, light transmission, fall velocity of dredged material, and the vertical distribution of dredged material in the water column following disposal. In addition, sub-bottom profiles and the overall bathymetry of the area were obtained to estimate the volume of material disposed of at the site.

Chemical studies were carried out on the water column and the sediments. Variables measured in the water column included temperature, turbidity, suspended solids, dissolved oxygen, pH, salinity, nutrients, PCBs, and heavy metals. Several approaches were employed in the measurement of sediment variables. These were bulk analysis, interstitial water, and elutriate tests. In all cases, PCBs and heavy metals were evaluated. Nutrients were analyzed only in interstitial water and during the elutriate testing. Bulk analyses included percent water, volatile solids, organic carbon, sulfides, Eh, pH, and oil and grease.

Bottom grabs were taken to characterize the types, abundance, and biomass of benthic organisms. Dermersal organisms were collected by trawling and were analyzed in terms of species composition, number/unit of effort, length, and weight for dominant finfish. Diet studies for finfish were also undertaken. The concentrations of PCBs and heavy metals in the tissue of fish and shrimp were determined to evaluate uptake and/or bioaccumulation of these substances. In addition, organisms were suspended in cages over the disposal mounds to examine toxicty and uptake of contaminants.

Findings

The dredged material was an oily, black, fine organic silt with a plastic texture. It was found to leave the disposal barge in clumps or as a well-defined mass and fall to the bottom with velocities of up to 180 cm/sec. Upon impact with the bottom, a dense surge of material flared outward at about 36 cm/sec and could be detected more than 200 m from the point of impact. Suspended solids returned to ambient conditions within 10 min, but a slight reduction in light transmittance persisted for several hours (18).

The disposal of 114,000 m^3 resulted in numerous mounds 2 to 3 m in height with a maximum radius of approximately 200 m. Subsequent chemical analyses for PCBs at 6 and 9 months after disposal indicated that the mound was gradually spreading (18). This movement was probably brought about by currents gradually redistributing the dredged material. The spreading was not of sufficient magnitude to move the contaminated sediments beyond the boundaries of the disposal site.

The majority of chemical changes in the water column during disposal were relatively minor (18). There were increases in dissolved manganese, ammonia, phosphorous, and total PCBs. These changes occurred with increases in suspended particulate matter, and, when particulate matter decreased, so did the concentrations of contaminants. The increase in particulate matter and associated chemical variables was of extremely short duration, usually less than 30 minutes. It is of interest that, prior to disposal, the concentrations of PCBs in the water column exceeded EPA criteria and these concentrations increased after disposal. It is possible that PCBs were entering Elliott Bay from the Duwamish Waterway and had approached equilibrium saturation values prior to disposal.

As would be expected, the chemical changes observed in the sediment are a reflection of the nature of the dredged material (18). Metals, nutrients, PCBs, and oil and grease were present in the disposal area sediment in greater concentrations after disposal than before disposal.

A number of biological variables were investigated during the Duwamish ADFI and a few showed major changes as a result of disposal. The number of species, density, biomass, and diversity of benthic invertebrates at the disposal site were depressed after disposal (when compared to predisposal values) (18). These effects were most apparent for the central stations of the disposal site and least noticeable for the corner stations. Some decreases in the above parameters were also noted at the two reference stations. Nine months after disposal the number of species present at the disposal site was comparable to the numbers present at the two reference sites although the biomass values continued to be depressed for the central and side stations of the disposal site. There was evidence that animals at the edges of the disposal site were stimulated by the dredged material.

As presented in Tables 13 and 14, there was essentially no uptake of metals or PCBs by fish or most invertebrates analyzed during and after the disposal operations. Pre- and post-disposal specimens were collected from the disposal site and locations outside Elliott Bay (18). In addition, caged animals were held at the disposal site for up to 3 weeks. Mussels held in cages at the disposal site accumulated PCBs to levels above background but the increase was not statistically significant. It should be pointed out, however, that some of the animals collected from Elliott Bay prior to disposal contained substantial amounts of PCBs so a slight uptake may not have been statistically significant.

TABLE 13

MERCURY AND CHROMIUM CONCENTRATIONS IN SEA CUCUMBERS

		Mercury, ppm		Chromium, ppm	
Exposure (weeks)	Sample	Disposal Site	West Reference	Disposal Site	West Reference
		Predisposal			
0	$\bar{X}$	0.01	0.01	0.32	0.32
		Postdisposal			
1	$\bar{X}$	0.01	0.01	0.26	0.24
2	$\bar{X}$	0.01	0.01	0.26	0.24
3	$\bar{X}$	0.01	0.01	0.26	0.24

MERCURY AND CHROMIUM CONCENTRATIONS IN SPOT SHRIMP

		Mercury, ppm		Chromium, ppm	
Exposure (days)	Sample	Disposal Site	West Reference	Disposal Site	West Reference
		Predisposal			
0	$\bar{X}$	0.06	0.06	0.64	0.64
		During Disposal			
3	$\bar{X}$	0.06	--	0.62	--
3	$\bar{X}$	0.06	--	0.61	--
7	$\bar{X}$	--	0.07	--	0.57

MERCURY AND CHROMIUM CONCENTRATIONS IN ALASKA PINK SHRIMP

		Mercury, ppm		Chromium, ppm	
Exposure (weeks)	Sample	Disposal Site	West Reference	Disposal Site	West Reference
		Predisposal			
0	$\bar{X}$	0.08	0.06	0.83	0.63
		Postdisposal			
2	$\bar{X}$	0.06	0.06	0.68	0.62
5	$\bar{X}$	0.07	--	0.63	--
14	$\bar{X}$	0.05	0.06	0.55	0.67
27	$\bar{X}$	0.04	0.05	0.50	0.58
39	1	0.04	0.05	0.62	0.70

TABLE 14

PCBs IN MUSSELS (*Mytilus edulis*) EXPOSED AFTER DISPOSAL

	Exposure (weeks)	Mean ppm
	0	0.122
West Reference Site:	1	0.103
	2	0.131
	3	0.100
Disposal Site:	1	0.108
	2	0.200
	3	0.206

PCBs IN SPOT SHRIMP (*Pandalus platyceros*) EXPOSED DURING DISPOSAL OPERATIONS

	Exposure (days)	Mean ppm
	0	0.174
West Reference Site:	7	0.190
Disposal Site:	3	0.208
	3	0.185

PCBs IN ENGLISH SOLE (*Parophyrys vetulus*)

	Sampling period	Mean ppm
West Reference Site:	Before Disposal:	2.28
	After Disposal:	
	2 weeks	0.65
	1 month	0.87
	3 months	*
	6 months	*
	9 months	5.90**
Disposal Site:	Before Disposal:	2.58
	After Disposal:	
	2 weeks	0.74
	1 month	*
	3 months	*
	6 months	*

* = No samples.
** = One sample only.

Demersal fish and shellfish (shrimp) seemed to ignore disposal. There were fewer present during disposal at the disposal site than at the east reference site but about the same number as at the west reference site. After disposal the number of fish decreased at the disposal site and at both reference sites; this decrease suggests a seasonal change in these organisms rather than an impact of disposal (18). The number of shrimp captured at the disposal site after disposal increased compared to those obtained prior to disposal. Shrimp at the reference sites either remained at the same level (east reference site) or increased erratically from month to month (west reference site). Overall, more shrimp were found at the disposal site after disposal than at either reference site, indicating that the shrimp were attracted to the disposal site.

SUMMARY

LABORATORY INVESTIGATIONS

Research results show that in general dredged material in the United States is not as toxic to aquatic organisms as originally conceived, based on bulk sediment analysis. Nevertheless, some sediments are toxic and disposal of these sediments may cause environmental harm.

DMRP Work Unit 1D06 (4) evaluated the possibility of obtaining a chemical extraction method for sediments which would reflect the availability of heavy metals to organisms. Studies to date have not produced such a technique, and there is no chemical method for environmental impact evaluation of dredged material prior to its disposal. DMRP Work Unit 1D11 (11) showed that although oil and grease levels could be high in sediments, a large part of what is routinely reported as oil and grease may be harmless elemental sulfur, and a large part of the hydrocarbon burden of sediments is not released from sedimentary particles nor is it available for gross uptake into the aquatic organisms tested.

Bioaccumulation by itself is difficult to interpret in terms of toxicity. Accumulation of a known toxicant in a human food source is of obvious importance. Components can be transferred through aquatic food chains with biomagnification. Accumulation may stress the organism and make it more susceptible to disease or predation. Necessary energy may have to be diverted into detoxification mechanisms. Lowered fecundity and abnormal larval development will ultimately have effects on species abundance and population dynamics within localized systems. These kinds of sublethal effects can culminate in an unexplainable population decline over an extended period of time.

FIELD INVESTIGATIONS

Freshwater site

There were but few important impacts as a result of dredged material disposal at the Ashtabula disposal site. Some chemical changes were observed, but these were of small magnitude and transient in nature. There were changes in the benthic community which persisted throughout the study; these primarily consisted of species replacement and an increase in the abundance of some organisms. Because these benthic organisms are of importance as food for fish, these changes would be of concern were it not that the feeding activities of fish in the area did not seem to be altered.

Estuarine Site

Disposal of material contaminated with PCBs in Elliott Bay during the Duwamish Waterway ADFI appeared to have a minimal impact. A disposal mound was created which gradually spread during the postdisposal period. There were minor changes in the chemistry of the water column. These appeared to be associated with a transitory increase in suspended particulate material, and, as soon as this material had settled, values for chemical parameters returned to predisposal conditions.

There was no significant uptake of PCBs or metals by organisms inhabiting the disposal area or by caged animals which were held in close proximity to the disposed material for up to 3 weeks. Some changes were noted in the abundance, diversity, and species composition of benthic invertebrates in the disposal area; however, similar changes in the reference area populations make it unlikely that disposal was wholly responsible for the changes.

APPLICATION AND REGULATION

The conceptual problem of toxicants associated with sediments must be evaluated in light of valid chemical and biological data describing the availability of toxicants to organisms and the water column prior to determining effects of such toxicants (2). Information must then be gained as to the effects of specific substances on organism survival and function. Many materials previously regarded as toxicants are not readily desorbed or released from sediment attachment and are thus less toxic than in the free or soluble state, on which most toxicity data are based.

Prater and Anderson (19), using a 96-hr bioassay technique with four different species or organisms, evaluated the toxicity of sediments from the Duluth, Minnesota-Superior, Wisconsin harbor. Sediments could be broadly classified on an arbitrarily selected scale as nonpolluted, moderately polluted, and heavily polluted using the bioassay. The results of an array of chemical analyses also led to an arbitrary designation of nonpolluted, moderately polluted, and heavily polluted sediments. In 75 percent of the cases chemical analyses supported bioassay results, but they were unable to identify the causal chemical factor for mortality. Concentrations of chemicals thought to be pollutants varied from one station to another and were not always highest at stations producing highest mortalities. Toxic properties of sediments could have been due to the action of one or more pollutants acting together (synergism) or to unidentified contaminants, particularly organic compounds. This strongly argues for the use of a whole-sediment bioassay to determine potential toxicity of dredged material for disposal (20). Although the suggested procedures have yet to be fully evaluated under a wide spectrum of environmental conditions, experience will undoubtedly validate this type of test over the long term.

There are now cogent reasons for rejecting many of the conceptualized impacts of disposed dredged material regarding potential toxicity based on classical bulk analysis determinations. It is invalid to use total sediment concentration to estimate contaminant levels in organisms since only a variable and undetermined amount of sediment-associated contaminants is biologically available. Although a few instances of uptake of possible ecological consequence have been seen, the fact that uptake depends on species, contaminants salinity, sediment type, etc., argues strongly that bulk analysis does not provide a reliable index of contaminant availability and potential ecological impact of dredged material.

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MANAGEMENT OF CONTAINMENT AREAS TO PROMOTE DEWATERING AND SOLIDIFICATION

C. C. Calhoun, Jr.
Environmental Laboratory
U.S. Army Engineer Waterways Experiment Station
Vicksburg, Mississippi 39180

ABSTRACT

Methods developed by the Dredged Material Resarch Program to manage dredged material containment areas to increase their service lives are discussed. Emphasis is placed on dewatering fine-grained dredged material by making maximum use of evaporation. Results of laboratory and a major field study are given. Methods of predicting dewatering attainable from evaporation are depicted.

BACKGROUND

In papers presented at the previous two meetings (1, 2), information was given on research being conducted as part of the U.S. Army Corps of Engineers' Dredged Material Research Program (DMRP) on methods to dewater/densify dredged material. These papers were prepared at a time when the research was in progress and consequently complete results, conclusions, and recommendations could not be given. The 5-year $33 million DMRP was successfully completed in March and more definitive information can now be given on the results of these studies.

The results of the DMRP are being published in over 200 technical reports. However, final results and guidelines from the program are being published in a series of 21 concise synthesis reports and most of the synthesis reports will also be published as Corps of Engineers' Engineering Manuals. Each synthesis report presents guidelines in one specific area addressed by the DMRP. One of these reports (3) represents the final DMRP guidelines on the management of disposal areas to maximize their service lives through dewatering/densifying the dredged material. As pointed out in the earlier papers (1, 2), the primary purpose of densifying the material is to gain the additional capacity of the site that would have been otherwise occupied by water. Since dewatering coarse-grained material and silts of low plasticity yield little or no additional volume reduction of the material, emphasis in the program was placed on dewatering fine-grained material. Although the specific purpose of dewatering (in the DMRP context) is to reduce the volume of and not to stabilize the material, dewatering usually results in the material having greatly improved engineering properties (e.g., higher strength, low compressibility, etc.). Consequently, in many cases, dewatering/densifying dredged material is synonymous with stabilizing the material.

GENERAL MANAGEMENT CONCEPTS

INTRODUCTION

Most concepts involving long-term disposal site management will result in increased capital construction and manpower/administrative cost. However, in most instances, the unit disposal cost of operating the site over the design life will be equal to or lower than cost for an unplanned operation. Thus, when incorporating the subsequently described procedures in an overall plan, the total cost, both short- and long-term, of concept implementation should be considered.

Application of any general concept to a given disposal site is a site-specific design problem. Whether or not any concept will prove feasible can only be determined after careful planning and study, not only of the concept itself, but of the alternatives to and constraints involved in concept implementation.

THIN LIFE PLACEMENT

The most economical dewatering mechanism is Mother Nature, through the evaporative process (4). The rates at which various soils will dewater through evaporation were studied in detail (5). The depth to which a layer of fine-grained dredged material will dewater is a function of the net evaporation at the site. The net evaporation is equal to the total evaporation over a given period of time minus the precipitation during that same period. In many areas the net evaporation is small and in fact is negative in areas where precipitation exceeds evaporation. The latter condition is common in many coastal areas.

In order for dredged material to dewater with no active dewatering techniques applied, the thickness of the lift placed in the containment area in general should not exceed the water loss available from about half of the net evaporation during the period from deposition to placement of the next lift. In some areas net evaporation available at particular times during the total period should be considered. For instance if the lift is to remain in the disposal area for one year before another lift is placed there may be several months where the net evaporation is considerably greater than the average for the entire year. During this time a desiccation crust on fine-grained dredged material will develop. Once a crust forms, subsequent wetting will not cause the crust to revert to the original near liquid state. Consequently, there may be crust formation and thus dewatering/densification in areas where the average net evaporation is low or negative.

It is recognized that in most instances it is not possible to place dredged material in extremely thin lifts because of the limited land area available for containing relatively large volumes of dredged material. In some cases the situation can be improved through such techniques as carefullly selecting the time of year (maximum net evaporation) for placement, the use of compartments within the area to increase the length of time the material will be in the area prior to placement of the next lift. These and other techniques are discussed in detail in References 3 and 6. In subsequent parts of this paper methods for taking advantage of rates approaching the total evaporation rate will be discussed.

OTHER MANAGEMENT CONCEPTS

Thin lift placement of fine-grained dredged material is the most economical management scheme when specific site conditions prmit. As man's input of energy into the system increases, costs also increase. As part of the DMRP, several techniques for dewatering fine-grained dredged material were considered and evaluated to varying degrees. The controlling factor in establishing feasibility was almost always economics. If the cost of dewatering the material exceeds the cost of constructing and using a new site, the dewatering technique is in general not feasible. However, this is a site-specific factor and what is not economically feasible in one area may be feasible in another area. In some instances, environmental concerns are paramount and costs are secondary to assuring the dredged material is contained in a specific area.

In the earlier papers (References 1 and 2), the dewatering techniques being considered in the DMRP were discussed in some detail and indepth discussion of all of the techniques will not be given here. Techniques considered were:

1. Progressive Trenching
2. Vacuum Wellpoints
3. Windmill Power Feasibility of Vacuum Wellpoints
4. Sand Slurry Injection
5. Natural Freezing and Thawing
6. Mechanical Crust Stabilization
7. Underdrainage
8. Low Voltage Gradient Electro-Osmosis
9. Vegetation Dewatering
10. Capillary Wicks

All of the techniques were successful to varying degrees. Details of the various studies and results are presented in References 7 and 8. The most universally applicable technique appeared to be progressive trenching where trenches are placed in the fine-grained dredged material to rapidly remove surface water plus increasing the effective evaporation rate.

EVAPORATION DRYING

During the early phases of the DMRP it was noted that a relatively thin crust usually developed over the underlying soft, wet fine-grained dredged material confined to a disposal area. It was assumed that this crust developed and retarded or eliminated any further evaporative drying by sealing the underlying material from the evaporative processes. It was noted, however, that in some instances the crust developed to relatively great depths and in some cases the crust would be as much as 15 feet thick (9).

A close evaluation of the site revealed that the confined disposal areas generally acted as "bath tubs" and water entering them through precipitation or other sources could not be removed except through evaporation. It was further noted that in areas where the net evaporation was high or where surface water was allowed to drain the crust thicknesses were much greater. As a result of these observations an extensive study of the mechanisms of crust development were made (5).

The laboratory and controlled field studies (5) revealed that the crust in fine-grained dredged material did not seal off the underlying soft material from the evaporative process. As the desiccation crust develops, cracks also develop. In most cases water would continue to be removed from these cracks by the evaporative process and the cracks will continue to develop as long as the water supplied by capillarity was less than the evaporative demand. Consequently, the desiccation cracks can extend to considerable depths and dewatering of the fine-grained dredged material will also continue to these depths.

When fine-grained dredged material is placed hydraulically in a containment area approximately four volumes of make-up water are required for a given volume of in situ sediment. Most of this water is drawn off the area through some type of sluice. In studies of fine-grained dredged material exhibiting various engineering properties, it was found that after the surface water was removed cracks would begin to develop as the water content of the material reached approximately 1.8 times the Atterberg liquid limit (LL) (5). This condition is referred to as the "decant point." The 1.8 LL is a statistical average and of course can vary from material to material.

As the desiccation cracks develop the average water content of the resulting polygons was found to be approximately 1.2 times the Atterberg plastic limit (PL). Again, the 1.2 PL is only a statistical average and may vary. As shown in Figure 1, the volume reduction through removal of water is linear and the slope of the straight line is the coefficient of shrinkage, C_s. The relation remains linear until the water content approaches the Atterberg shrinkage limit (SL). The desiccation cracks were found to account for as much as 20 percent of the total volume change. The rate of water loss through evaporation was found to vary depending on whether the dredged material was derived from marine or freshwater sediments. For marine or saltwater dredged material the estimated water loss in inches, centimeters, etc., is equal to about 35 percent of the effective evaporation rate while for freshwater the loss is approximately 50 percent of the effective evaporation rate. In order to take maximum advantage of evaporative drying, surface water must be removed. This includes removal of surface water that is held in the desiccation cracks. A technique for removing surface water was developed by the DMRP and will be discussed in detail in this paper.

PROGRESSIVE TRENCHING

BASIC CONCEPT

The desiccation cracks formed in fine-grained dredged material are interconnecting and thus a system of trenches within a disposal area will remove

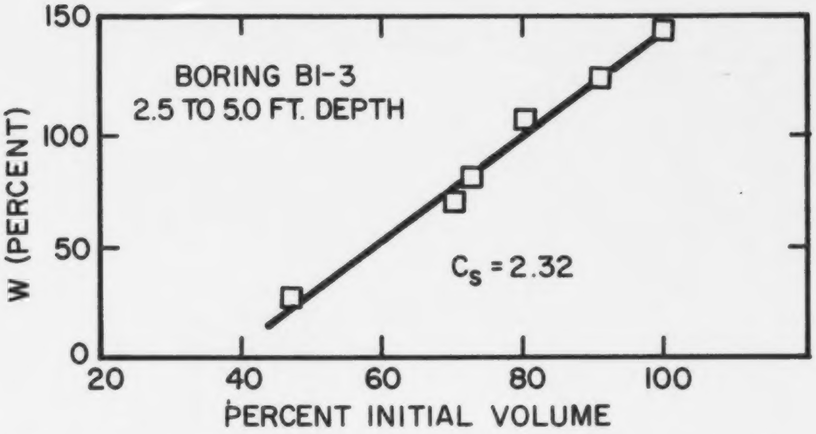


Figure 1. Typical linear shrinkage curve for dredged material tested (from Reference 4).

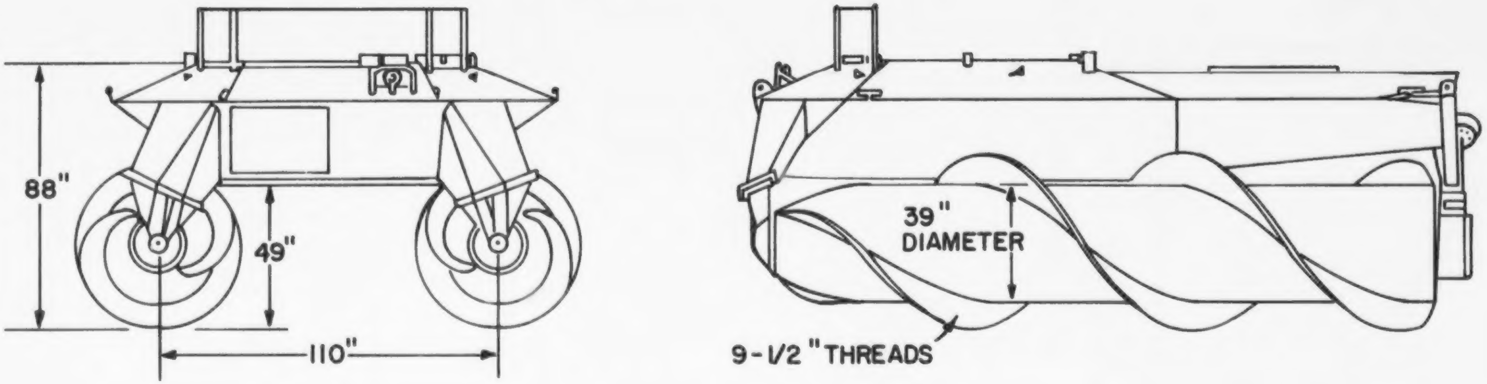
water from both the surface and from the cracks if the flowline of the trenches is below the bottom of the cracks. Since the material underlying the crust is at or above its LL a trench much deeper than the crust thickness cannot be maintained. Consequently, as the desiccation cracks deepen the trenches must also be progressively deepened to assure the water in the cracks will be drained.

The progressive trenching concept lowers the surface elevation of the dredged material through three mechanisms. The first is through shrinkage of the upper material in the crust. Second, the progressive lowering of the perched water table in the containment area increases the effective stress acting on the soft underlying dredged material thereby causing it to consolidate. The third mechanism depends on the foundation material and water table conditions. If the foundation material is relatively soft and the perched water table connects with the foundation water table, the increased effective stresses from lowering the perched water table will cause the foundation to consolidate.

TRENCHING TECHNIQUES

The major problem associated with the progressive trenching concept is the construction of the trenches themselves. At the decant point the material is too soft to support conventional equipment such as draglines. Equipment for operation in disposal areas was evaluated as part of the DMRP (10). Only one piece of equipment was found that could operate in the soft dredged material and rapidly produce trenches. The Riverine Utility Craft or RUC (Figure 2) was found to be suitable if not ideal for use. The RUC produces trenches with the tracks formed by the rotors. The RUC was described in detail in an earlier paper (2).

In many respects the RUC is similar in appearance and operational characteristics with the Dutch-made Amphirol. However, the RUC is a much heavier vehicle and the rotors are over three times larger, making the RUC a much more effective trenching tool. The Dutch use the Amphirol in the "ripening" proc-



RIVERINE UTILITY CRAFT

EMPTY WT. (INCLUDING DRIVER AND FUEL)	11,000 POUNDS
GROSS WT. (DRIVER, FUEL AND PAYLOAD)	13,000 POUNDS
LENGTH, FT. (OVERALL)	246/12 = 20.5
WIDTH, FT. (OVERALL)	168/12 = 14
HEIGHT, FT. (OVERALL) (LESS WINDSHIELD)	92 /12 = 7.67
ROTOR SPACING, CENTER TO CENTER, IN.	110
ROTOR DIAMETER (DRUM ONLY) IN.	39
ROTOR DIAMETER (OVER HELIX) IN.	58
ROTOR LENGTH (OVERALL) IN.	222
ROTOR LENGTH (IN CONTACT WITH GROUND, NO RUT) IN.	195
GROUND CLEARANCE, IN.	49
FLOATING DEPTH (EMPTY) IN. (WATER)	21.5
FLOATING DEPTH (LOADED) IN. (WATER)	24.0

Figure 2. Specifications of Riverine Utility Craft (RUC).

ess of dredged material placed in polders (11). The ripening process is similar to the progressive trenching technique.

In the progressive trenching technique initial shallow trenches are made after the material reaches the decant point. The trenches are progressively deepened as required with the RUC until the trenches are about two feet deep, the limit of the RUC's effectiveness. With two feet of crust other more conventional equipment can be selected using procedures outlined in Reference 10 to continue the deepening process. Details of the trenching techniques are given in Reference 3.

FIELD TESTS

The progressive trenching technique was field evaluated at the Upper Polecat Bay (UPB) disposal site in Mobile, Albama (4). The 85-acre site filled with 8 to 12 feet of fine-grained dredged material was described in Reference 2 along with the general plan of tests.

Computations were made based on extensive data collected at UPB to predict the dredged material surface settlement or volume gain in the site. The perched water table at the UPB site did not connect with the foundation water table. Consequently foundation consolidation was not considered and because of the soft nature of the material, rebound of the foundation was not expected and did not occur as the perched water table was lowered. Containment area volume gain was expected only from shrinkage of the crust and dredged material consolidation.

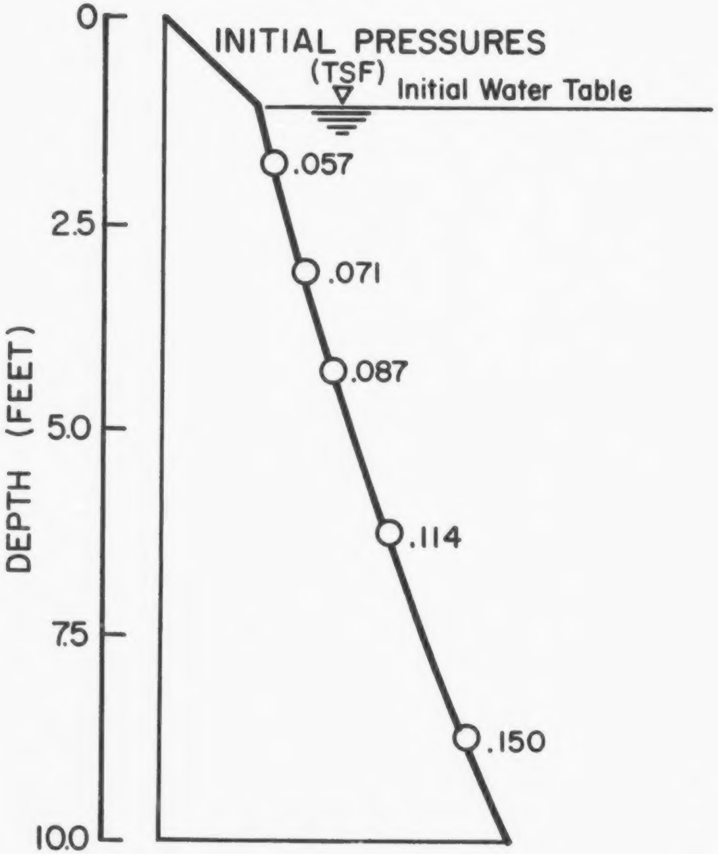
Average data used to predict shrinkage and consolidation are shown in Figure 3. The settlement due to shrinkage was determined from the following relationship:

$$\Delta H = H \frac{\Delta w}{C_s} \tag{1}$$

Where ΔH = Settlement due to shrinkage.
H = Thickness of strata.
Δw = Average change in water content, percent.
C_s = Coefficient of shrinkage.

Since the dredged material had been placed several years prior to initiating dewatering, the water content of the material was at about its LL. Also, data indicated that average water content of the polygons would be about at the PL (as opposed to the statistical average of 1.2 PL discussed earlier). Δw was then taken to be equal to LL-PL. The average C_s was 2.34. Predicted shrinkage as a function of drawdown is shown in Figure 4.

Conventional theories were used to compute expected settlement from consolidation of the soft dredged material (4). The resulting consolidation as a function of drawdown is shown in Figure 5. The coefficient of consolidation of the material was computed to be 0.3 cm^2/sec which indicates that dredged material consolidation would reach 90 percent of total primary consolidation approximately two months following imposition of load.



H (ft)	e_1	C_c	C_s	LL %	PL %	PI %
1.25	3.45	1.20	3.35	123	39	83
1.25	2.92	1.01	2.06	83	29	53
1.25	2.83	1.01	2.06	83	29	53
2.50	2.30	0.78	2.12	81	29	52
2.50	2.17	0.75	2.26	80	28	52

Figure 3. Average consolidation and shrinkage data for settlement analysis.

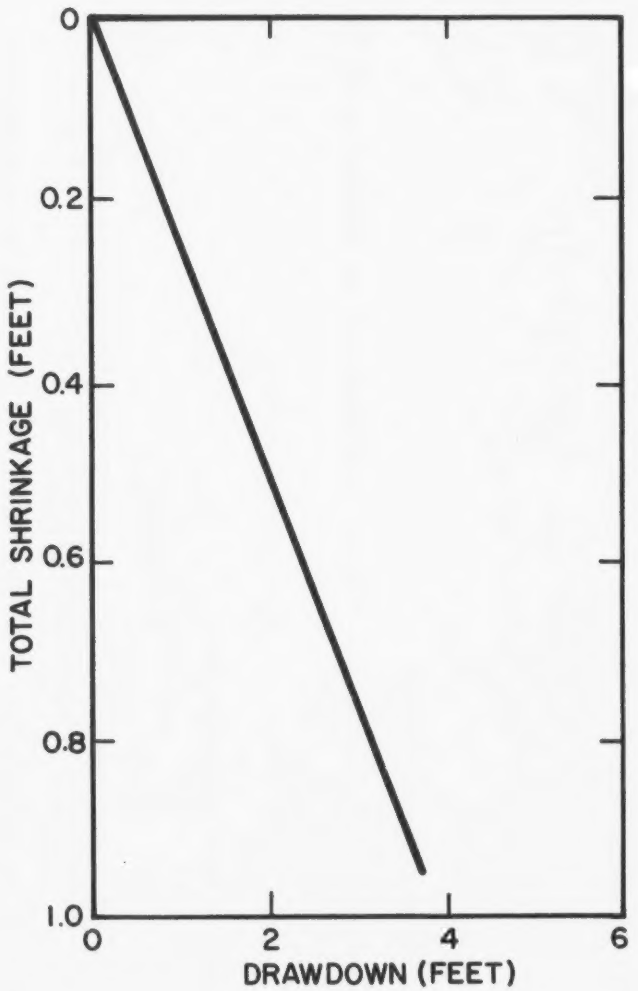


Figure 4. Potential dredged material shrinkage versus drawdown.

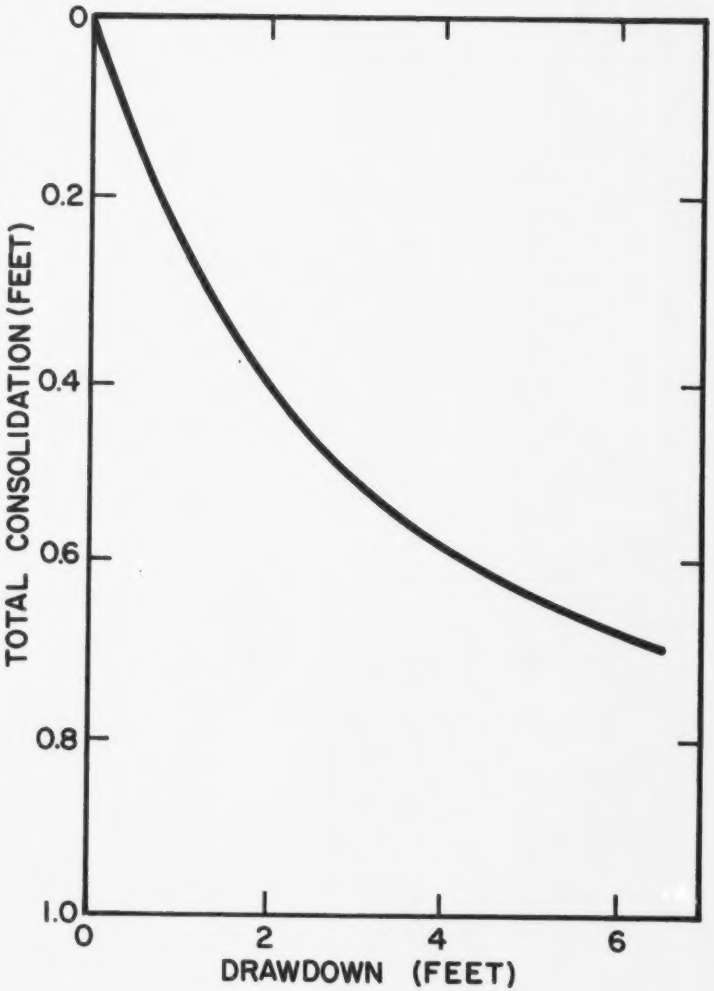


Figure 5. Potential dredged material consolidation versus drawdown.

Total predicted settlement is shown in Figure 6. Because of the relatively short time required for the material to consolidate, total settlement was the sum of that from shrinkage and consolidation at a given drawdown. Field data (also plotted on Figure 6) show that the predicted settlements were less than those observed in the field. The difference in the predicted and observed behavior was not great and the predicted values were on the conservative side. Therefore, the analytical technique appears acceptable.

PREDICTION EQUATIONS

The following equations have been developed to predict dewatering effects once the dredged material has reached the decant point. The approximate thickness, in feet, of dredged material at the end of the decant phase H_{dm}, when surface drainage improvement should be initiated, is given by

$$H_{dm} = 27V_{cs}\left\{\frac{\frac{\%\ \text{sand}}{100} + \left[\left(1 - \frac{\%\ \text{sand}}{100}\right)\frac{w_{cd}}{w_{cs}}\right]}{43{,}560A_{ds}}\right\} \tag{2}$$

where V_{cs} = volume of channel sediment to be dredged, cu yd.
% sand = percentage of sand in the channel sediment.
w_{cd} = average water content of the dredged material at the end of the decant phase, percent.
w_{cs} = average water content of the channel sediment, percent.
A_{ds} = area of the disposal site, acres.

In lieu of better data, especially if the calculation is made prior to dredging, w_{cd} may be taken at 1.8 x LL, and Equation 2 may be written as

$$H_{dm} = 27V_{cs}\left\{\frac{\frac{\%\ \text{sand}}{100} + \left[\left(1 - \frac{\%\ \text{sand}}{100}\right)\frac{1.8\ LL}{w_{cs}}\right]}{43{,}560A_{ds}}\right\} \tag{3}$$

Effect of evaporative dewatering

The estimated water loss, in inches, from evaporative dewatering ΔW is given by the relationship

$$\Delta W = 0.35E_p \tag{4}$$

for saltwater dredged material and by

$$\Delta W = 0.50E_p \tag{5}$$

for freshwater dredged material, where E_p is the Class A Pan Evaporation (see Reference 3) for the dewatering interval.

The depth, in inches, to which the initial thickness of dredged material will be dewatered H_i is given by

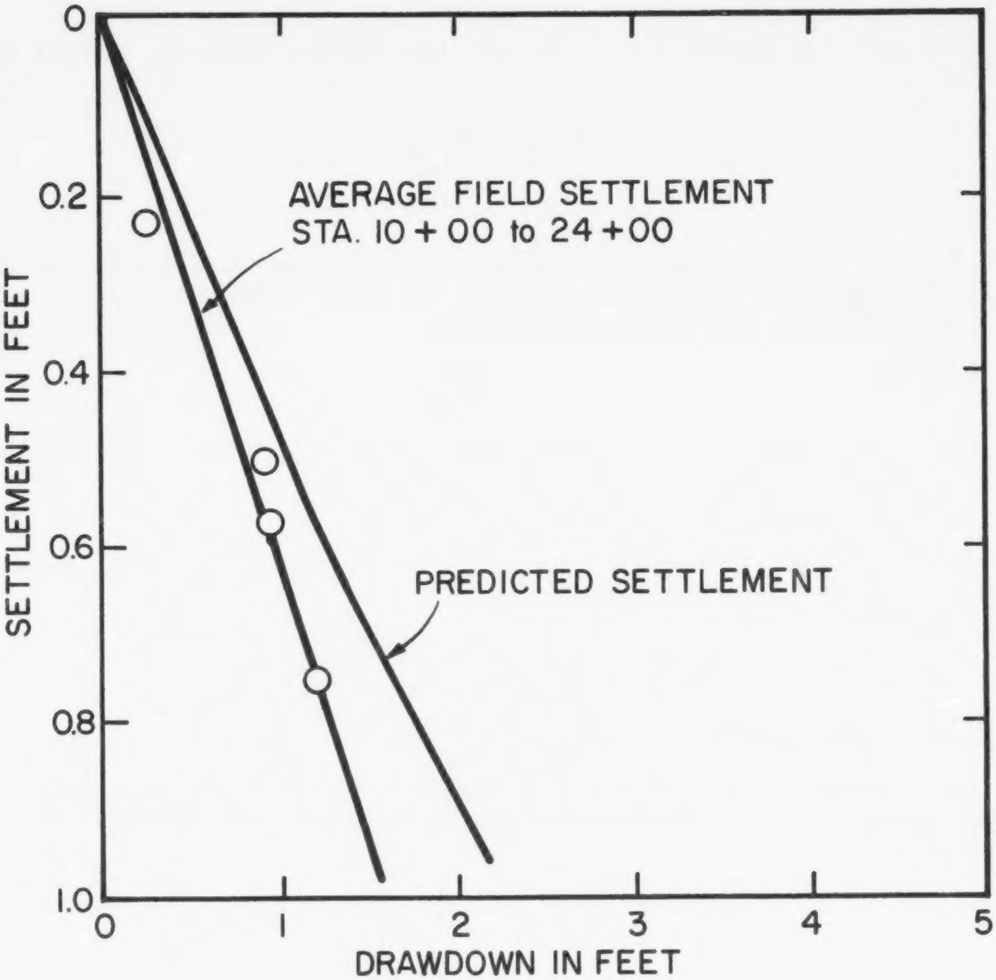


Figure 6. Potential settlement versus drawdown and field results.

$$H_i = \frac{w_{cd}}{w_{cr}}\left[\frac{\Delta W}{(\frac{w_{cd}}{w_{cr}}) - 1}\right]\left(1 + \frac{1}{0.01 w_{cd} G_s^2}\right) \qquad (6)$$

where w_{cr} = average water content of the dewatered dredged material (crust), percent.

G_s = specific gravity of the channel sediment/dredged material solids.

In the absence of better data, w_{cd} may be assumed as 1.8 x LL, while w_{cr} may be assumed as 1.2 x PL. In such a case, Equation 6 becomes

$$H_i = 1.5\,\frac{LL}{PL}\left[\frac{\Delta W}{(1.5\frac{LL}{PL} - 1)}\right]\left(1 + \frac{1}{0.018 LL G_s^2}\right) \qquad (7)$$

Vertical subsidence, in inches of the dredged material surface H_s is computed as

$$H_s = \frac{\Delta W}{1 + \frac{P_{dc}}{100}} \tag{8}$$

where P_{dc} is the percentage of total dewatering volume gain due to the volume of the space between desiccation cracks. In lieu of better data, P_{dc} may be taken as 20 percent, and Equation 8 becomes

$$H_s = \frac{\Delta W}{1.2} \tag{9}$$

The crust thickness, in inches, formed by desiccation H_{cr} is given by

$$H_{cr} = H_i - H_s \tag{10}$$

The estimated volume gain, in cubic yards, from evaporative dewatering shrinkage V_{gd} is computed by

$$V_{gd} = A_{ds} \Delta W \left(1 - \frac{\%\ \text{sand}}{100}\right) \frac{43{,}560}{12(27)} \tag{11}$$

Effect of increase effective stress consolidation

The average initial effective stress, in pounds (force) per square foot, at the center of the undewatered dredged material (subcrust) layer P_i is estimated as

$$P_i = \left[\frac{G_s(1 + 0.01w_{cd})}{1 + 0.01w_{cd}G_s} - 1\right] \frac{\gamma_w}{12} \left(\frac{H_{dm} - H_i}{2} + H_i\right) \tag{12}$$

where γ_w is the unit weight of sale or freshwater, in pounds (force) per cubic foot. In lieu of better data, w_{cd} may be taken as 1.8 x LL, and Equation 12 becomes

$$P_i = \left[\frac{G_s(1 + 0.018LL)}{1 + 0.018LLG_s} - 1\right] \frac{\gamma_w}{12} \left(\frac{H_{dm} - H_i}{2} + H_i\right) \tag{13}$$

The increase in effective stress, in pounds (force) per square foot, from water table lowering Δp is approximated by

$$\Delta p = H_i \gamma_w \tag{14}$$

The approximate consolidation settlement, in inches, resulting from increased effective H_c is approximated by

$$H_c = \frac{(H_{dm} - H_i)C_c}{1 + 0.01w_{cd}G_s} \log \frac{p_i + \Delta p}{p_i} \tag{15}$$

where C_c is the compression index for the dredged material. If better data are not available, w_{cd} may be taken as 1.8 x LL, and Equation 15 becomes

$$H_c = \frac{(H_{dm} - H_i)C_c}{1 + 0.018LLG_s} \log \frac{p_i + \Delta p}{p_i} \tag{16}$$

Additional disposal volume gain, in cubic yards, from subcrust consolidation V_{gc} is computed from

$$V_{gc} = H_c A_{ds} \left(1 - \frac{\%\ \text{sand}}{100}\right) \frac{43{,}560}{12(27)} \tag{17}$$

Total settlement, in inches, of the dredged material surface from dewatering H_t is thus

$$H_t = H_s + H_c \tag{18}$$

Total disposal area volume gain, in cubic yards, from dewatering V_{gt} is given as

$$V_{gt} = V_{gd} + V_{gc} \tag{19}$$

The thickness, in inches, of subcrust remaining to be dewatered H_r is given by

$$H_r = H_{dm} - H_i - H_c \tag{20}$$

The volume, in cubic yards, of dredged material available for removal and productive use V_p is estimated as

$$V_p = A_{ds} \left\{ \left[H_{cr} \left(1 - \frac{\%\ \text{sand}}{100}\right) \right] + H_{dm} \frac{\%\ \text{sand}}{100} \right\} \frac{43{,}560}{12(27)} \tag{21}$$

More detailed information on the use of these equations and other required computation is given in Reference 3 which also contains example problems. A computer program has been developed to assist with the analyses (12).

SUMMARY

The proper management of disposal areas can increase the service lives of areas containing fine-grained dredged material. Guidelines have been developed during the DMRP to aid in developing management plans (3, 13). Work will continue under the Corps of Engineers' Dredging Operations Technical Support program to verify and refine the methodologies.

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IMPACTS OF OIL SPILL AND CLEAN-UP ON THE EUROPEAN COAST: AMOCO CADIZ

William P. Davis
Chief, Bears Bluff Field Station
Gulf Breeze Environmental Research Laboratory
U.S. Environmental Protection Agency
Wadmalaw Island, S.C.

INTRODUCTION

Accidental release of petroleum or other hazardous substances during transport or transfer from ships will continue to be a major focal point of pollution control and abatement efforts. At the time of this writing the Amoco Cadiz wreck on the Brittany coast of France (Figure 1) represents the largest oil spill resulting from man's actitivies. It occurred on one of the world's most productive sea coasts in a nation where both technological and scientific skills abound. Heavy weather contributed to both the causes for the wreck and the subsequent spread of the petroleum to over 200 linear kilometers of biologically rich, high energy coastal habitats. From the moment the French realized the existance of the impending disaster there was a public mobilization of fishermen, farmers, students, scientists, and clean-up equipment; all attempting to take action appropriate to prepare for the coming of the "black tide" or "maree noire".

This report attempts to summarize highlights of impacts, ecological effects, and evolving response efforts of interest to persons living or working along the shores of the world's oceans.

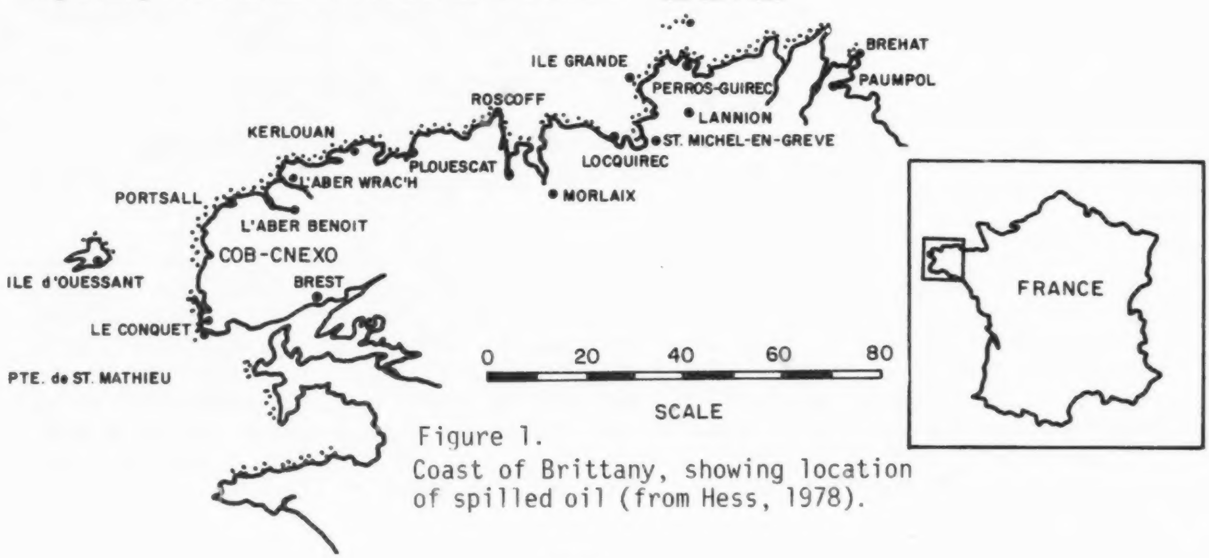
LES SEPT ILES
BREHAT
ILE GRANDE
PAUMPOL
PERROS-GUIREC
ROSCOFF
LANNION
KERLOUAN
ST. MICHEL-EN-GREVE
PLOUESCAT
LOCQUIREC
L'ABER WRAC'H
PORTSALL
MORLAIX
L'ABER BENOIT
COB-CNEXO
ILE d'OUESSANT
BREST
FRANCE
LE CONQUET
0
20
40
60
80
PTE. de ST. MATHIEU
SCALE

Figure 1.
Coast of Brittany, showing location of spilled oil (from Hess, 1978).

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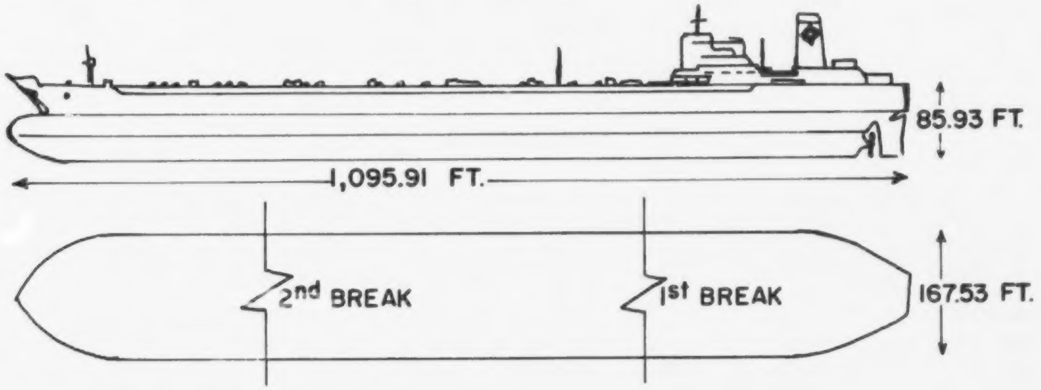
CHRONOLOGY OF SPILL EVENTS

The Amoco Cadiz (Figure 2) hit rocks one mile (1.6 km) offshore from the town of Portsall, France on 16 March 1978. The ship's steerage had been lost and attempts to pull her from the rocks failed as several tow lines parted. Winds of 40-50 knots contributed to these failures; within six hours the ship began breaking up, resulting in the release of 216,000 tons of Iranian and Arabian light crude oil, plus 4,000 tons of fuel into the ocean. During the next week heavy seas emulsified the escaping petroleum into a stinking "mousse" and spread the spill over 200 kilometers of the Brittany coast.



REGISTRY: MONROVIA LIBERIA
OWNER: AMOCO TRANSPORT CO.
CALL/OFF Nº.: A8AN/4773
POWER: 2SA 8CYL. DIESEL: 980mm × 2,000mm
22.678 Kw/30.400 BHP. (SINGLE SCREW)
SPEED: 15.25 KTS.
MAX. DRAFT: 65 FT./19.81 M.
CAPACITY: 233.690 DWT.
109.700 GRT.
91,000 NET/TONS

ABOVE DATA PER LLOYDS REGISTER.

TANK LAYOUT AND VESSEL DRAWINGS NOT AVAILABLE.

Figure 2. General layout and dimensions of Amoco Cadiz showing locations of breaks (from Hann et al., 1978).

Geologically, the Brittany coast is a shield shoreline, underlain by igneous and metamorphic rock complexes, with headlands, crenulate bays, small and large beaches, rocky islands, tidal flats, marshes, all representing widely varied ecological habitats. Although no major rivers enter the area, a number of "abers" or small rivers are found. Initially, wind moved the petroleum in an easterly direction piling oil up against every westward facing coastal feature, or concentrated it into estuaries with westward facing mouths or entrances. These westerly winter winds are part of the annual weather pattern for this part of the ocean, Manche, also known as the English Channel.

In early April winds typically swing around and become northeasterly. The shift in wind direction occurred about 3-4 April, 1978 (Figures 3a,b and 4a, b). The shift redistributed much petroleum to previously shielded sections of the Brittany coast.

In addition, the highest spring tides of the season (9-meter tide range) occurred within the two-week span after the wreck (Figure 5). This resulted in wide exposure of coastal intertidal habitats to oil, necessitating considerable ingenuity by the people of Brittany in attempting to clean their impacted coast.

The Ministry of Interior and the French Navy directed much of the effort to disperse oil at sea with chemical agents. Little information is available on just what chemical dispersants were employed, where they were used or in what quantities. Apparently over 50 tons of dispersants were sprayed during 18-22 March. Four ships were observed spraying sheen, not mousse, on 8 April 1978. These ships had been making daily sorties since 18 March from the port of Brest, France. Interviews with petroleum company officials revealed that at least four different chemical agents, plus chalk, had been applied but no useful documentation was available.

The NOAA/EPA Special Report: "The Amoco Cadiz Oil Spill - A Preliminary Scientific Report" (Hess, 1978) provides a valuable summary of initial events, responses and assessment of the spill. Subsequently, a French-United States special commission has been established, co-chaired by Dr. Lucien Laubier, Director of Centre Oceanologique de Bretagne (COB) of the Centre National pout l'Exploration des Oceans (CNEXO), and Dr. Wilmot N. Hess, Director of NOAA Environmental Research Laboratories (ERL). This commission meets each six months to review scientific activities for assessing effects of the spill, and to award funds for specific research grants to consider the long-term effects of petroleum from the spill.

Oil spills are a challenge to pollution-oriented ecologists. Even after a number of international symposia which have produced extensive reports (Bates, 1978), one gets the impression that the ecologist faces a desperate struggle when oil spill catastrophes occur. The magnitude of these events, particularly from ships the size of Amoco Cadiz is simply beyond the scale of experience of most scientists and planners. Furthermore, the emergency atmosphere surrounding clean-up operations has rarely incorporated much consideration or realization of need for scientific assessment. The Amoco Cadiz oil spill has spurred efforts in the United States to initiate ecological impact assessment combining physical oceanography, coastal dynamics, and habitat assessment, ecological/behavioral effects with chemical analyses. Multinational teams brought together in Brittany by the Amoco Cadiz incident have been developing and exchanging newly created ideas (Conan et al. 1978), techniques and strategies to use scientific methods for impact assessment, estimation of damage to marine ecosystems and eventually, procedures for habitat restoration.

Events following the Amoco Cadiz oil spill have provided quantities of material for the press, sociologists, economic strategists, and politicians. With regard to the ecology, the assessment of effects still continues.

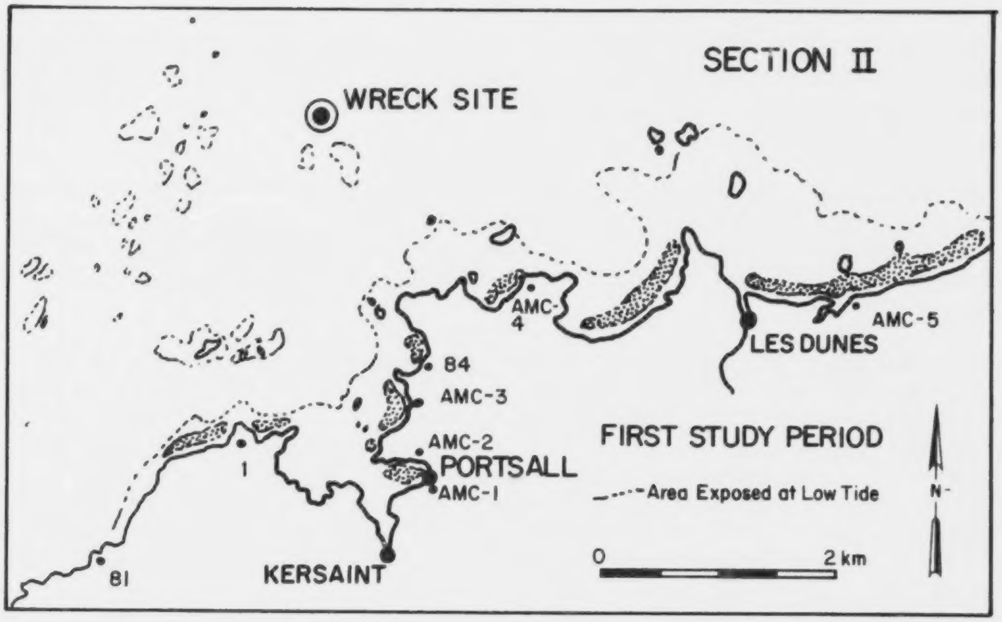

SECTION II
WRECK SITE
AMC-4
AMC-5
LES DUNES
84
AMC-3
AMC-2
PORTSALL
AMC-1
1
81
KERSAINT
FIRST STUDY PERIOD
Area Exposed at Low Tide
0
2 km
N

Figure 3a. Locations of observation stations in Section II, the Portsall area, during the first study period (March 19 to April 2). Heavy oil accumulations are indicated by the dark-stippled pattern.

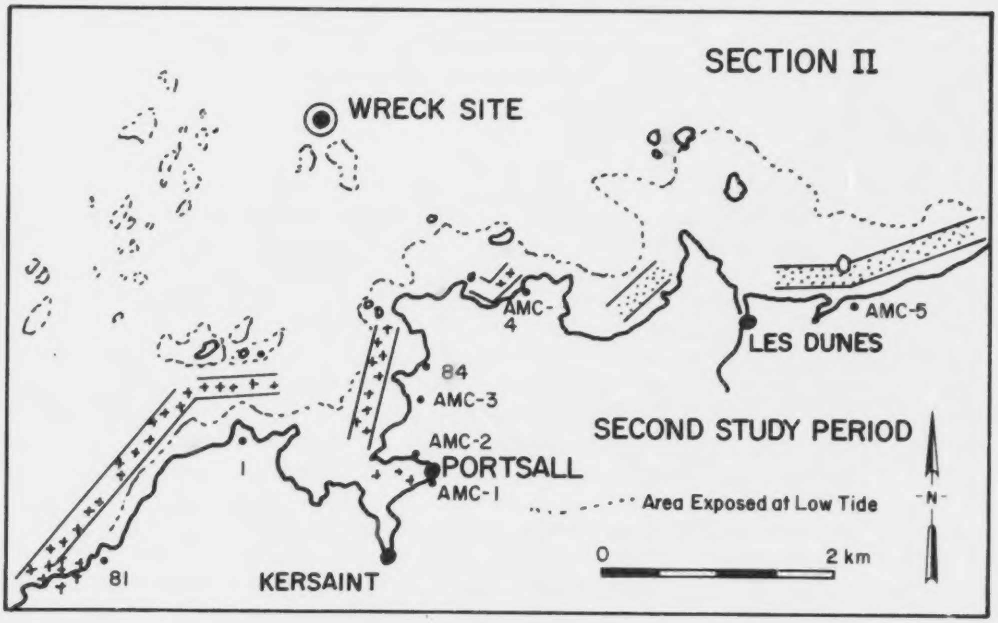

SECTION II
WRECK SITE
AMC-4
AMC-5
LES DUNES
84
AMC-3
AMC-2
PORTSALL
AMC-1
1
81
KERSAINT
SECOND STUDY PERIOD
Area Exposed at Low Tide
0
2 km
N

Figure 3b. Oil distribution for Section II for second study session, April 20 to 28. Heavy and light oil coverage are indicated by the plus and light-dot patterns, respectively (from Gundlach and Hayes, 1978).

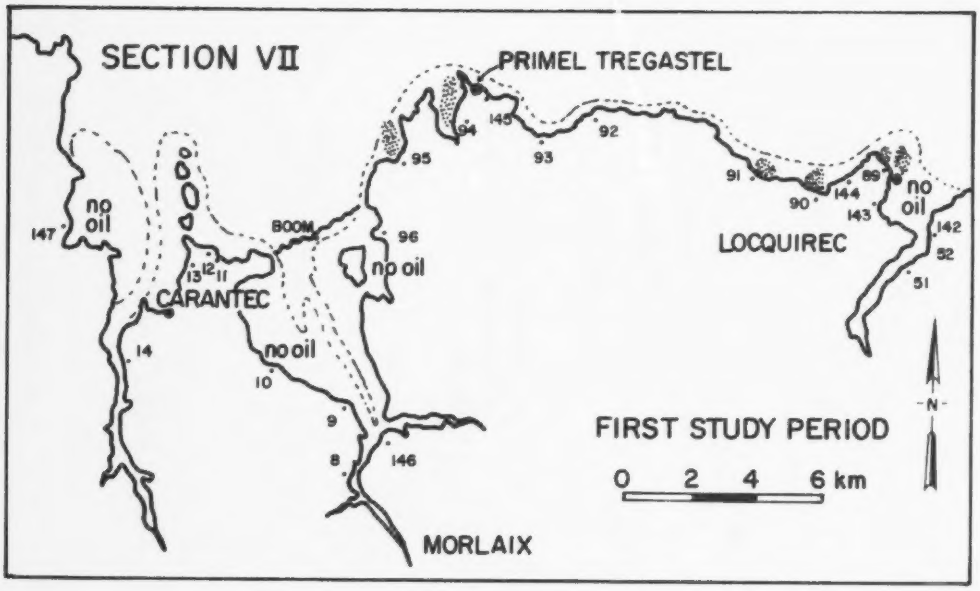

SECTION VII
PRIMEL TREGASTEL
no oil
no oil
no oil
no oil
no oil
BOOM
CARANTEC
LOCQUIREC
MORLAIX
FIRST STUDY PERIOD
0 2 4 6 km
N
147
13 12 11
14
10
9
8
146
96
95
94
145
93
92
91
90
144
143
89
142
52
51

Figure 4a. Location of stations in Section VII, Roscoff to Pointe de Plestin (F-142). Oil distribution during the first study session, March 19 to April 2, is indicated by the dark-stippled pattern.

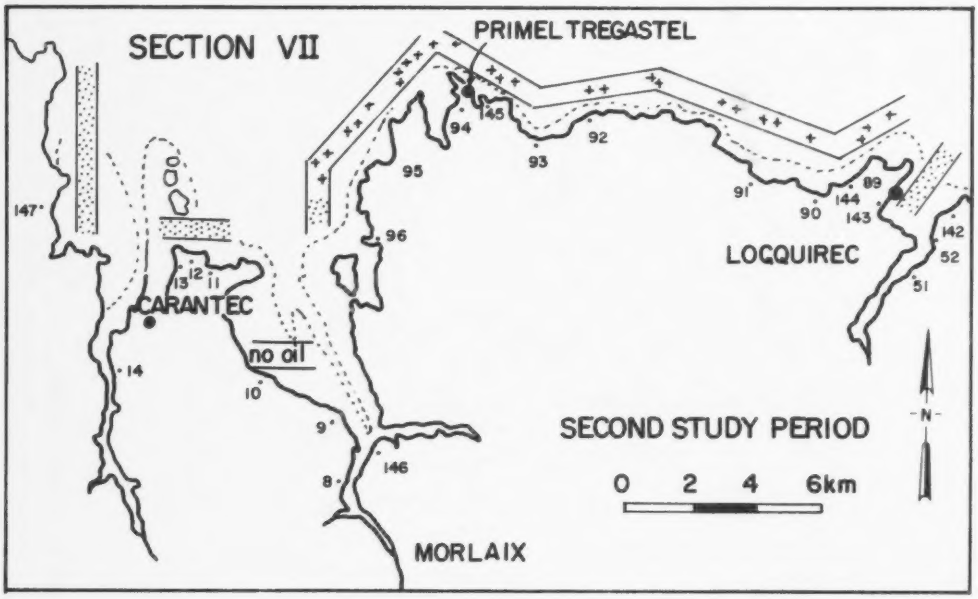

SECTION VII
PRIMEL TREGASTEL
CARANTEC
no oil
LOCQUIREC
MORLAIX
SECOND STUDY PERIOD
0 2 4 6km
N
147
13 12 11
14
10
9
8
146
96
95
94
145
93
92
91
90
144
143
89
142
52
51

Figure 4b. Oil distribution along the coastline of Section VII during the second study session, April 20-28. Heavy and light oil coverage are indicated by the plus and light-dot patterns, respectively (Gundlach and Hayes, 1978).

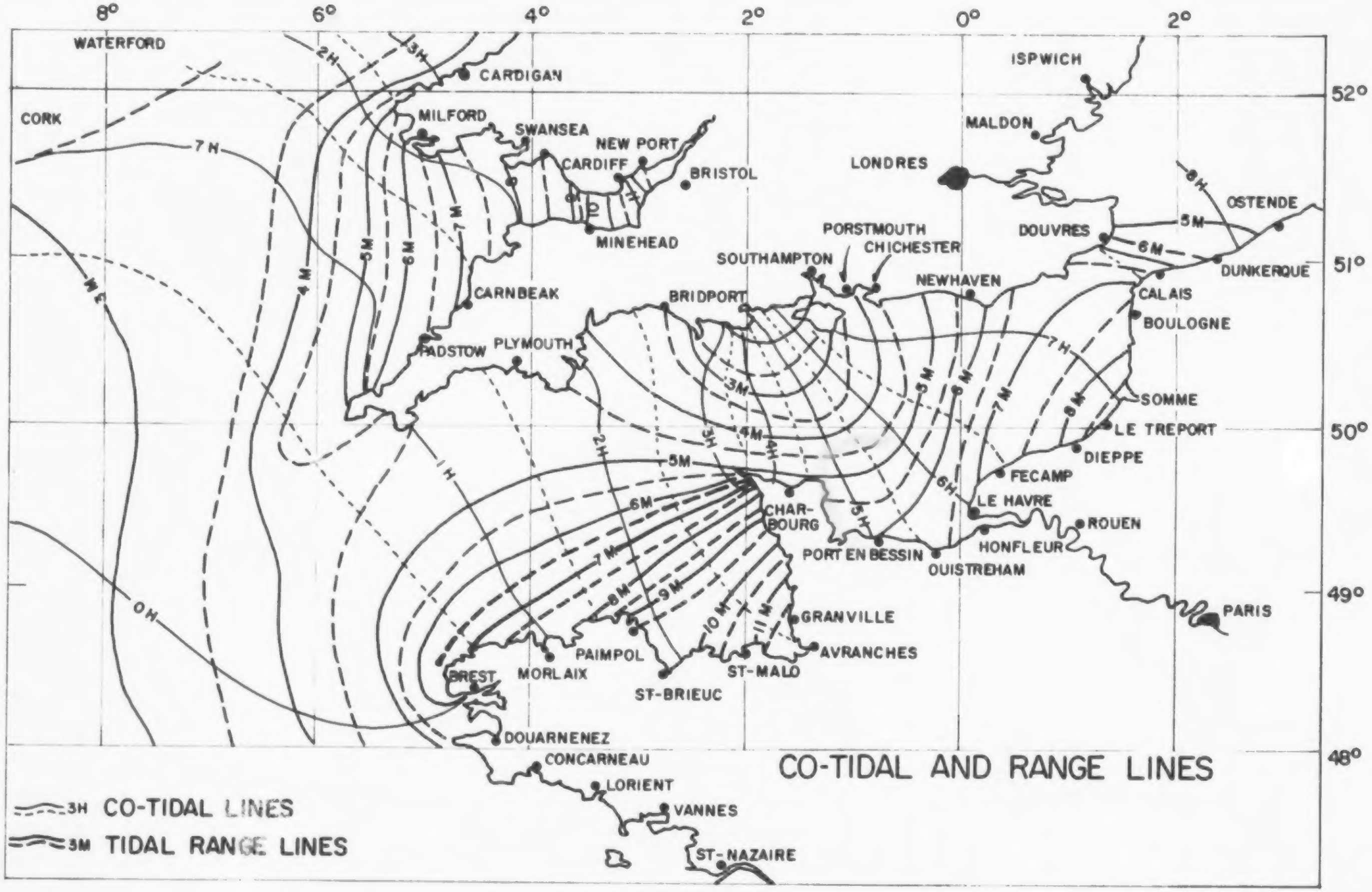

8°
6°
4°
2°
0°
2°
52°
51°
50°
49°
48°
WATERFORD
CORK
CARDIGAN
MILFORD
SWANSEA
NEW PORT
CARDIFF
BRISTOL
MINEHEAD
CARNBEAK
PADSTOW
PLYMOUTH
BRIDPORT
SOUTHAMPTON
PORSTMOUTH
CHICHESTER
NEWHAVEN
LONDRES
MALDON
ISPWICH
DOUVRES
OSTENDE
DUNKERQUE
CALAIS
BOULOGNE
SOMME
LE TREPORT
DIEPPE
FECAMP
LE HAVRE
ROUEN
HONFLEUR
OUISTREHAM
PORT EN BESSIN
CHAR-BOURG
GRANVILLE
AVRANCHES
ST-MALO
ST-BRIEUC
PAIMPOL
MORLAIX
BREST
DOUARNENEZ
CONCARNEAU
LORIENT
VANNES
ST-NAZAIRE
PARIS
7H
3H
2H
1H
0H
4H
5H
6H
8H
2M
3M
4M
5M
6M
7M
8M
9M
10M
11M
CO-TIDAL AND RANGE LINES
3H CO-TIDAL LINES
3M TIDAL RANGE LINES

Figure 5. Tidal range lines for the English Channel (from French Hydrographic Service Pub. 551). Tidal range increases, going from 6 m at Brest to over 9 m at Paimpol (Gundlach and Hayes, 1978).

BIOLOGICAL-ECOLOGICAL EFFECTS

Initially, some 4000 or more birds perished, including puffins, quillemots, and razorbill auks which are considered endangered species in this region. Additionally, cormorant species (shag), loons and other migrating diving birds were severely affected. Certain intertidal organisms including species of molluscs, echinoderms, crabs, worms, algae, crustacea, and fishes were also killed.

In one case, oil contamination of the shoreline caused mass death of sea urchins (Echinocardium) and several associated species of molluscs. In this particular situation it was evident that organisms which are normally burrowing forms had surfaced, apparently driven from the substrate by petroleum components which penetrated up to 30 cm into the sandy bottom habitats. Whether this event is related to use of dispersants or storm-induced mixing of seawater, sediment and petroleum is not yet understood.

Bretagne benthic organisms have been studied for a number of years. The amphipod crustaceans comprise 90% of the species diversity and 40% of the biomass in fine sediments. Normally amphipods are present in numbers of 15,000/m^2 in summer, and 8,000/m^2 in winter. Within a fortnight of the wreck, the population dropped to 2-5/m^2, the number of species from the normal 23 to only 7 (Cabioch, personal communication; NOVA, 1979). These organisms represent the principal food for commercial fishes including sole, sea bream and others. Molluscs and polychaete worms at these particular stations were not so dramatically affected.

A calcareous algae, Lithothamnion, forms "brush" piles, or areas with high amounts of interstitial space. Such habitats have become benthic entrapments for oil-sediment mixtures. This phenomenon has not been extensively studied and needs careful evaluation to determine potential ecological impacts.

Dr. Claude Chasse´ has conducted extensive benthic biological surveys at Morlaix, and other bays including the Bay of Lannion, where in the aforementioned example early effects of oil and/or dispersants to benthic organisms was most dramatic. His research (Chasse´, 1972) spans ten years of study of the macrobenthos between St. Efflam and Michel-en-Grève. At this site, 3-7 km of beach were covered with five windrows of sea urchins, razor clams and other organisms. Continuation of ecological assessments at this site, Lannion, Morlaix, and others are of extreme importance to better understanding of long-term effects of petroleum on benthic organisms.

Through the University of West Brittany, students were organized into survey cadres. During initial phases of the spill some 150-200 students visited intertidal areas (during low spring tide) and counted benthic organisms using printed survey sheets. These sheets were collected and compiled by Drs. Chasse´ and Glémarec. More than 600 students took part in repeating the "body counts" of dead organisms. The results of these repetitive visits and subsequent assessment of benthic populations will be invaluable to ecological comprehension of oil-organism interactions and mechanisms.

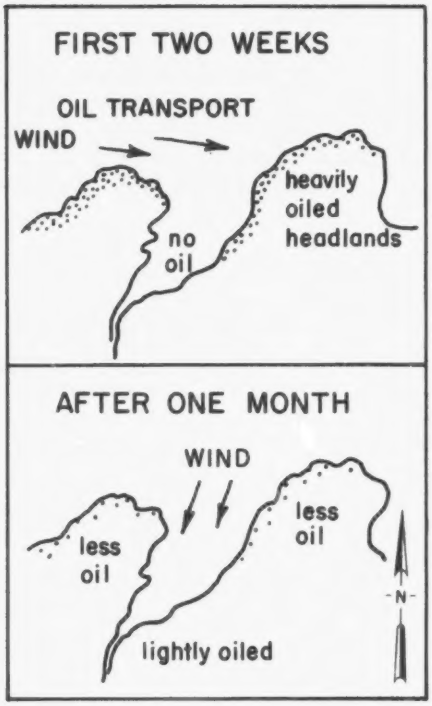

FIRST TWO WEEKS
OIL TRANSPORT
WIND
heavily oiled headlands
no oil
AFTER ONE MONTH
WIND
less oil
less oil
lightly oiled
N

Figure 6. (Top) Oil pushed by strong westerly winds during the first two weeks was mainly deposited along westerly-facing headland areas. Interior embayments generally remained free of oil. (Bottom) A wind shift during the beginning of April spread a light layer of oil deep into the embayments. Previously deposited oil along the exposed headlands was greatly reduced in quantity (Gundlach and Hayes, 1978).

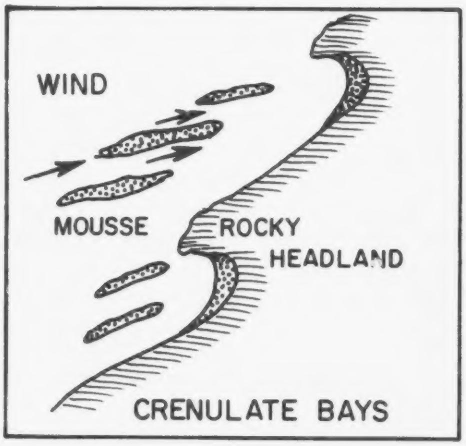

WIND
MOUSSE
ROCKY HEADLAND
CRENULATE BAYS

Figure 7a. Entrapment of oil by crenulate bays. Generally, the southerly section of each bay remained free of oil.

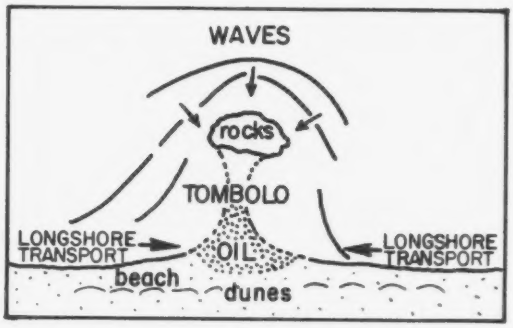

WAVES
rocks
TOMBOLO
LONGSHORE TRANSPORT
OIL
LONGSHORE TRANSPORT
beach
dunes

Figure 7b. Illustration of the tombolo effect causing localized oil deposition behind offshore rocks (Gundlach and Hayes, 1978).

Many small marsh areas were impacted, but the one surrounding the embayment south of Isle Grande represents one of the most heavily damaged. Here the configuration of the coast entrapped oil and winds forced the oil onto a *Juncus* and *Spartina* marsh (Figures 6 and 7). Oil depth over the marsh surface varied with specific circumstances, but was as deep as 15 cm in some areas. Effects on biota were devastating; polychaetes and other mobile invertebrates littered isolated pools of water by the thousands.

Clean-up of marsh areas was given high priority. However, use of heavy equipment without restriction to single-access work paths further contributed to destruction of the soils and additional disruption of the habitat. Relative effects of oil versus clean-up activities will be difficult to assess. The area poses considerable challenge to planners interested in restoration.

The emotional impact of any oil spill is symbolized by the tragedy of wild bird deaths. In the case of *Amoco Cadiz*, the eastward extension of the spill lashed Les Sept Iles (Seizh Enez) (Figure 8) which represents the southernmost nesting colony of the common puffin, an endangered species. The observation team was escorted to the sanctuary on 2 April by ornithologists from the Ligue Francaise pour la Protection des Oiseaux (LPO). At sites where one would normally see approximately 300 pairs of puffins in prenesting courtship, only one bird was found. By 14 April "bird hospitals" had recorded over 850 puffin carcasses. Spring migration was still underway so the total impact of oil on the Sept Isles colony is unclear. Other bird species using this sanctuary include razorbills (100 pairs), guillemots (300 pairs), kittiwakes (100 pairs), storm petrels, herring, greater- and lesser- black-backed gulls, fulmars (100 pairs) and gannets (4,000 pairs).

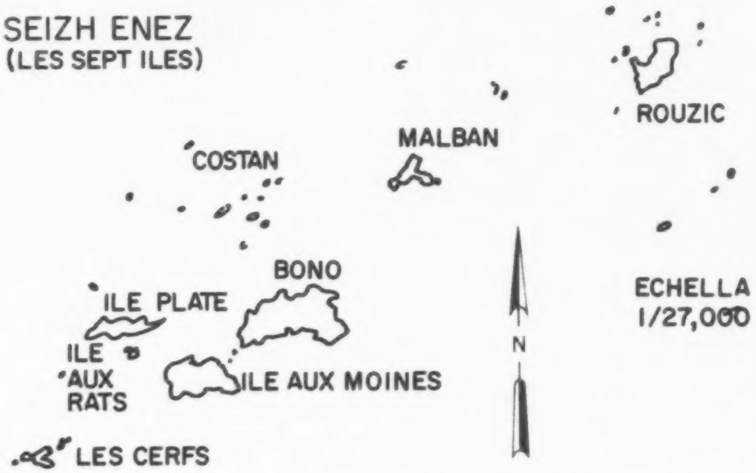


Figure 8. Islands constituting Les Sept Iles (from Milon, 1972).

The species most affected by oil were diving birds (puffins, razorbills, cormorants, loons); soaring and far-flying species (gulls, gannets) had far fewer immediate deaths. The word *immediate* is emphasized to denote acute toxicity. In the case of gulls, these birds were observed feeding on weakened molluscs, crabs, and worms. Gannets were seen carrying seaweed back to their nesting sites from areas affected by oil slicks. It is not known whether contaminated seaweed can cause the kind of mortality which has been demonstrated experimentally with duck eggs (Albers, 1977) or gull eggs (Patten and Patten, 1977). By 2 April, three dead gray seals had been recovered from this area of the Brittany coast.

The impact on birds was noted to be "less than expected" by personnel working for LPO and Societe pour l'Etude et la Protection de la Nature in Bretagne (SEPNB). This observation reflects the important role of specific circumstances. Did many birds avoid this spill because it was so massive and contained highly aromatic petroleum components? Were migration patterns sufficiently different during this year (compared to Torrey Canyon)? Had populations (as known for puffins, razorbills, and guillemots) already been decreased by the Torrey Canyon spill, so there were fewer birds to be affected? Answers to these questions will depend on research and better understanding, evaluation and knowledge of population dynamics.

FISHERIES AND MARICULTURE

Husbandry and harvesting of the sea is as traditional as farming the land in Brittany. Coastal waters of the North coast affected by the spill support a traditional artisan seaweed industry. Many species of algae are harvested and commercially processed to produce alginates for food, soap and other products. Other algae, as well as the Laminaria, are harvested by hand (up to 6-7 thousand metric dry tons/year) and used as a supplement to livestock food.

Much of the contaminated algae was removed during clean-up activities, since these plants acted as natural mops for intertidal oil. The impact upon the algae industry cannot be assessed with the present data, although harvesting was being carried out during the summer of 1978. It initially appears that washing by sea action prevented much of the damage.

Rearing of oysters (Crassostrea gigas and Ostrea edulis) is an important industry in Brittany. A valiant effort was initiated at the onset of the spill to place numbers of oyster racks in estuaries southwest of the anticipated trajectory of the floating oil. Of course there was insufficient time to remove all the racks. Total assessment of the impact of oil pollution on oysters is not yet available.

Certain oyster culture areas of Brittany had initially been considered "spared" from the fate suffered by such westward facing bays as Aber Wrach or Aber Benoit. Later, Morlaix oysters became petroleum tainted and considered unfit to eat, even though the surface mousse never came to that area.

Mussel culture is also practiced, but the effects of the Amoco Cadiz oil spill have not yet been assessed. Part of the problem is related to the distribution and effects of the oil-sediment mixture which has been transported by tidal currents to areas not significantly impacted by the surface mousse (e.g., Morlaix). The benthic oil poses particular threats to crab, scallop, and commercial fish species, emphasizing the need for comprehensive chemical-ecological studies of the food webs of coastal fishery populations.

Another issue is whether traditional fishery population study techniques can adequately discern effects of a massive pollution event such as a major oil spill. One gets the impression, from both the literature and from discussions with statisticians, that present methodology does not yet meet these kinds of ecological assessment needs.

AGRICULTURE

Another impact which emphasizes the environmental interrelationship of sea, atmosphere, and land is the unassessed effect of airborne contaminants. On Easter morning, the stench of petroleum was evident 10-20 km downwind from the coast with the onshore winds blowing at 40-50 knots. Fields rich with spring crops, including mustard greens, broccoli and cauliflower extended to the edge of the rocky coast and it was apparent that fields were "browned" from petroleum-laden mist blown in from the stormy seas. It is hoped that studies were made to assess the impact on this link in the food chain.

VULNERABILITY OF BRITTANY COASTAL HABITATS

On the basis of experience in the study of earlier petroleum spill, i.e., Metula (Straits of Magellan) and Urquiola (Spain), Gundlach and Hayes (1978) and Gundlach (1979) have classified coastal habitats into a vulnerability index designed to predict longevity of oil in specific habitats. In this system, on the scale of 1 to 10, the greater the value, the greater the likelihood of long term damage. Table 1, taken directly from Gundlach and Hayes (1978), includes comments addressing the situation one month after the wreck.

OIL SPILL CLEAN-UP ACTIVITIES

In June, 1977, legislation had been adopted in France to prevent and control marine pollution from accidents or releases of pollutants. The plan had become known as "Plan POLMAR" (from "plan contra pollution marin"). Authority from the government of France flows from the national to local level via bureaucratic networks. Control of pollution was conceived by "Ministry of the Quality of Life", but executed under "Direction of Equipment" which is comparable to Departments of Public Works or Corps of Engineers as far as operational aspects are concerned. Enforcement authority was retained by the Ministry of the Interior. The first major spill to test Plan POLMAR was Amoco Cadiz. The maritime prefect at Brest was placed in charge of POLMAR; but the operational approach was split into marine and coastline responsibilities due to the "two fronts" requiring action.

Hann et al. (1978) summarized the first two months of clean-up activities directed at Amoco Cadiz petroleum. Probably no other spill has received so thorough a clean-up, nor been so completely assessed (Figure 9,10 and Appendix).

Over 5,000 French troops were mobilized for shoreline clean-up. Because of the Easter holidays thousands of students on spring vacation volunteered for clean-up and ecological survey teams; engineer (road and construction)

TABLE 1. THE OIL SPILL VULNERABILITY INDEX WITH PARTICULAR REFERENCE TO THE AMOCO CADIZ OIL SPILL[1]

Vulnerability Index	Shoreline Type; Example	Comments
1	Exposed rocky headlands; Douarnenez to Pte. du Raz and Primel-Trégastel to Locquirec	Wave reflection kept most of the oil offshore; no clean-up was needed.
2	Eroding wave-cut platforms; south of Portsall and F-1 to F-82	Exposed to high wave energy; initial oiling was removed within 10 days.
3	Fine-grained sand beaches; stations south of Roscoff (AMC-9 and 10) and east of Portsall (AMC-5)	All only lightly oil-covered after one month, mainly by new oil swashes.
4	Coarse-grained sand beaches; AMC-stations 4 (near Portsall) and 12 (St. Cava) and F-38	Oil coverage and burial after one month remained at moderate levels.
5	Exposed, compacted tidal flats; La Grève de St. Michel	No oil remained on the sand flat but did cause the enormous mortality of urchins and bivalves.
6	Mixed sand and gravel beaches; no really good example of this beach type	The index value is due to rapid oil burial and penetration; all areas had compacted subsurface which inhibited both actions.
7	Gravel beaches; stations F-80 95 and 129, also AMC-16	Oil penetrated deeply (30 cm) into the sediment; clean-up by use of tractors to push gravel into surf zone seemed effective and not damaging to the beach.
8	Sheltered rocky coasts; common throughout the study area	Thick pools of oil accumulated in these areas of reduced wave action; clean-up by hand and high pressure hoses removed some of the oil (this process is valid in non-biologically active areas).
9	Sheltered tidal flats; behind Ile Grande and at Castel Meur	Tidal flats were heavily oiled; clean-up activities removed major oil accumulations but left remaining oil deeply churned into the sediment; biological recovery has yet to be determined.
10	Salt marshes; Ile Grande marsh	Extremely heavily oiled with up to 15 cm of pooled oil on the marsh surface; clean-up activities removed the thick oil accumulations but also trampled much of the area; biological recovery has yet to be determined.

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[1] Higher index values indicate greater long-term damage by the spill. For further information, consult Hayes, Brown, and Michel (1976) or Gundlach and Hayes (1978).

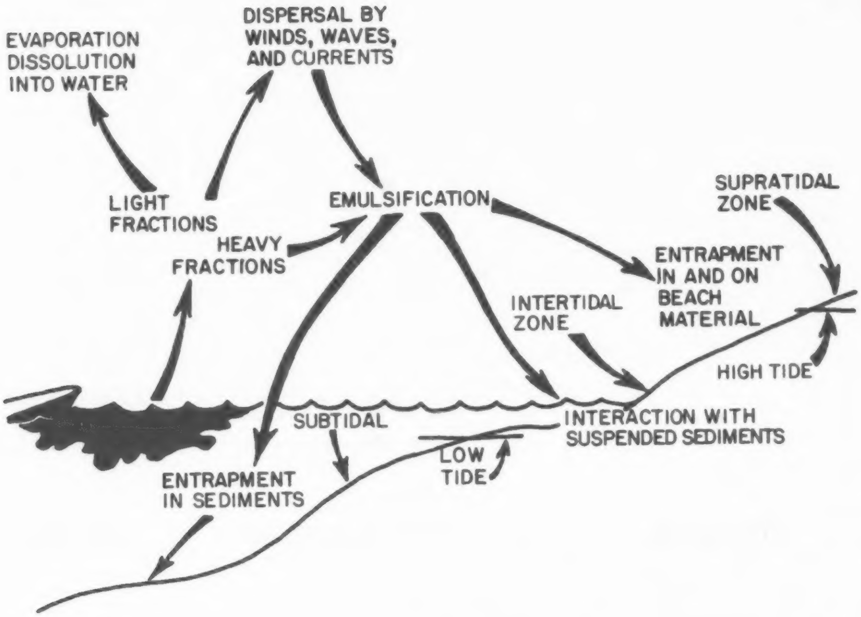

EVAPORATION
DISSOLUTION
INTO WATER
DISPERSAL BY
WINDS, WAVES,
AND CURRENTS
LIGHT
FRACTIONS
EMULSIFICATION
SUPRATIDAL
ZONE
HEAVY
FRACTIONS
ENTRAPMENT
IN AND ON
BEACH
MATERIAL
INTERTIDAL
ZONE
HIGH TIDE
SUBTIDAL
INTERACTION WITH
SUSPENDED SEDIMENTS
LOW
TIDE
ENTRAPMENT
IN SEDIMENTS

Figure 9a. Mass balance components (Hann *et al.*, 1978).

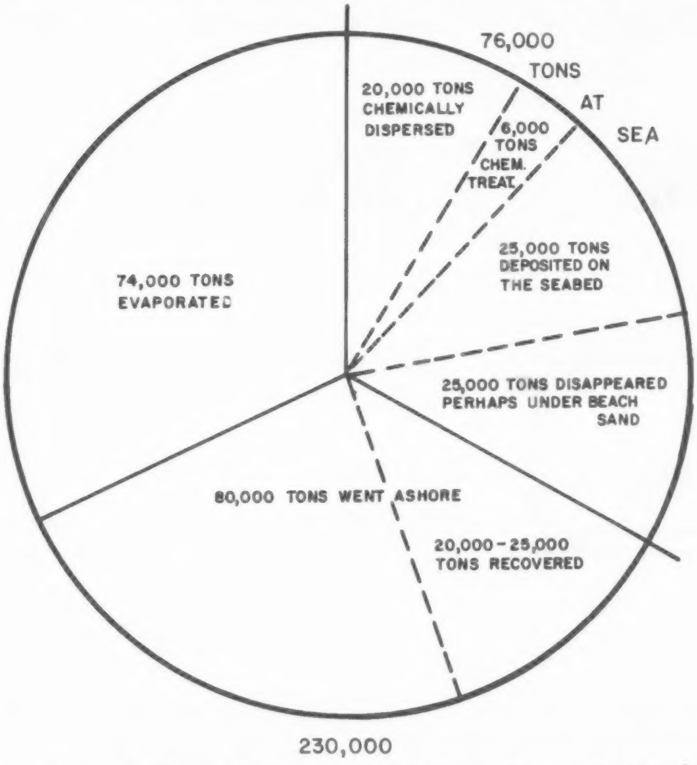


Figure 9b. Estimated fate of oil from *Amoco Cadiz* (Source: local newspapers; from Hann *et al.*, 1978).

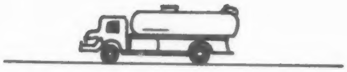
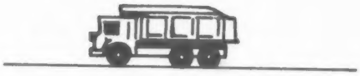
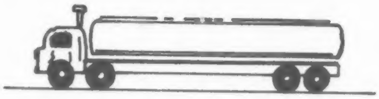
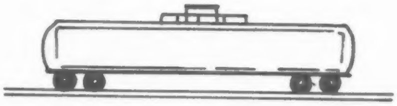
	VOLUME OF LOAD	LOADS TO MOVE MOUSSE
	750 GALLONS	188,830
	1,800 GALLONS	78,678
	252 CU. FT. 1,884 GALLONS	75,170
	4,000 GALLONS	35,405
	8,000 GALLONS	17,702

Figure 10. Equivalent truck and tank car loads necessary to remove potential volume of mousse from shore (Hann *et al.*, 1978).

cadres and contractors also assisted in the clean-up effort. One of the most spectacular responses came from the farmers of Brittany with their intrepid tractors and liquid manure or "honey" wagons. This equipment was particularly well suited for use in the sandy-rocky coastal areas, for pumping mousse from wherever it became entrapped, such as in coves, pools, or from man-made entrapment basins or oil booms. Hann et al. (1978) estimated that approximately 190,000 loads of about 3,000 liters each of oil/water emulsion were moved by "honey wagon" to collection points for transport by truck, tanker, or railroad to tankers at the port of Brest. Tankers carried the oil/water emulsion to refineries. Tragedies continued as the small coastal ship, Henrietta Bravo, sank in heavy seas while transporting 900 tons of oil-contaminated algae which had been arduously removed from the shoreline of Brittany.

Land activities eventually became divided into four zones, each under the responsibility of an engineer from Direction of Equipment, with reinforcements from the Army. Student volunteers were under administrative direction of the Ministry of Youth and Sports. An effort was made to enlist volunteers who had adequate protective clothing, shelter, good personal health, and identification accountability for assignment to specific cadres under trained foreman. Such details may seem remote from the focus of this report, but they are mentioned to indicate the inadequacy of existing systems to respond to spillages of the magnitude of Amoco Cadiz. There are serious problems in dealing effectively with well-meaning volunteers; advance planning and training are essential.

Approaches to oil clean-up spanned every possibility. Rapid deterioration of the ship, combined with weather problems, precluded effective removal of the cargo at sea. High waves acted as a surge pump, emptying the cargo into the waters. Inflatable booms were deployed along the coast, but the incredibly heavy seas, high winds and 8-meter tides overwhelmed these devices as protective mechanisms. However, the booms did concentrate the oil and help in controlling it for pumping and removal efforts.

Some downwind areas anticipated the petroleum influx and bulldozed sand on tourist beaches above the intertidal zone. After the clean-up, beach sand was graded over the cleaned zone.

Petroleum dispersants were applied offshore with the restriction that these chemicals not be used in waters less than 50 meters deep. This arbitrary depth was probably unrealistic since 50 meters is actually very close to shore which, combined with the proximity of the slick and onshore winds, aggravated this strategy of oil spill control and mixed much petroleum into highly productive coastal waters. Thirty-five boats were ultimately used to spread dispersants north and east of Roscoff and Ile de Brehat areas.

Onshore oil was pushed, scraped, sponged, collected in ditches, collected mixed with algae and wrack resulting in immense disposal and storage problems. On 6 April, after the change of wind direction, aerial observations revealed that the volume of floating and entrapped petroleum in the environment was overwhelming, even compared to the vast clean up efforts being expended. It has been stated that over $70 million (U.S.) was spent by June 1978 and it would be at least two years more before "important" areas were completely cleaned.

In the case of village shorelines, as exemplified at Roscoff, limited detergents and high pressure water spraying was used to clean rocks. Observations on 21 July revealed evidence of the oiling by both the numerous tar spots and a notable decrease of molluscs and marine organisms seen at the earlier (27 March) observations. In July there was an extensive growth of the alga Enteromorpha which is often associated with nutrient or biostimulant pollutats in coastal marine waters. Bathing beaches along Roscoff were being scraped and raked and truck loads of the algae hauled away to reduce the organic decomposition odors and blockage to water access for beach visitors.

Other high energy beaches looked fresh with clean deposits of sand. Trenches dug 20-60 cm down invariably revealed a deposit layer of brown mousse, still pungent and sticky. These oil layers will likely be reexcavated during winter tide cycles and oil will be redistributed by tidal and surface currents during winter 1979.

The Amoco Cadiz oil spill presented to modern technology, to man's society and the ecosystem an awesome and formidable challenge: how to protect from, or mitigate the effects of very large oil spills. Unlike some other spills, there were high levels of science, technology, and organization available at this time and place.

Concentrated study and consideration of all aspects of this event will help prepare us for the possibility of future accidents. It is clear that the combinations of circumstances and possible impacts are nearly infinite. Yet, it is also clear that numerous preparations and steps can be taken to improve safeguards for both prevention and response.

ACKNOWLEDGMENTS

This summary is compiled from the work of a number of investigators. Through the generous and patient support of Dr. Lucien Laubier, Director of the Centre Oceanologique de Bretagne, and many members of his staff including Dr. G. Conan, Dr. L. d'Ozouville, and Mr. P. de Clarens, it was possible to study a delicate and complicated situation. Dr. J. Bergérard, Director of the Station Biologique, 29211, Roscoff, generously assisted. Dr. Louis Cabioch of Station Biologique spent considerable time with the observation team, shared data, showed undersea video tapes and provided much insight and research knowledge on the oil-sediment emulsion and effects on benthic communities. Dr. Miles O. Hayes, Jacqueline Michel Hayes and Erich Gundlach generously shared their time, valuable observations, and data derived from many hours of study of coastal ecosystem dynamics. The National Oceanographic and Atmospheric Administration's Environmental Research Laboratory Director, Dr. Wilmot N. Hess is thanked for facilitating the author's second visit to Brittany and the opportunity to participate as an observer at the Amoco Cadiz Commission meeting at Brest. I wish to acknowledge the assistance of my patient spouse who contributed invaluable aid in the preparation of the manuscript. Ms. Sharon Maier and Douglas Middaugh read and made improvements to the manuscript.

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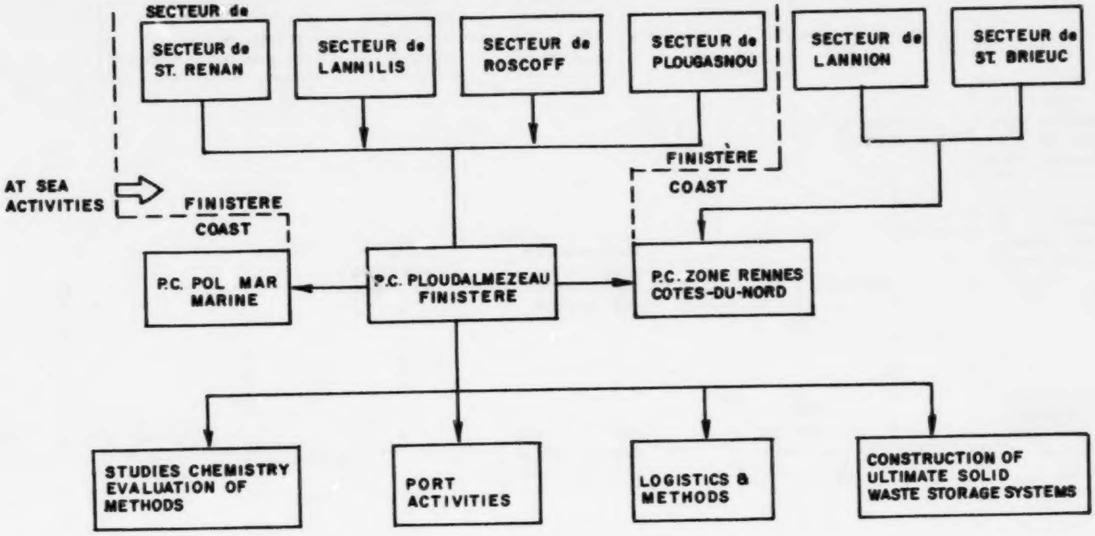
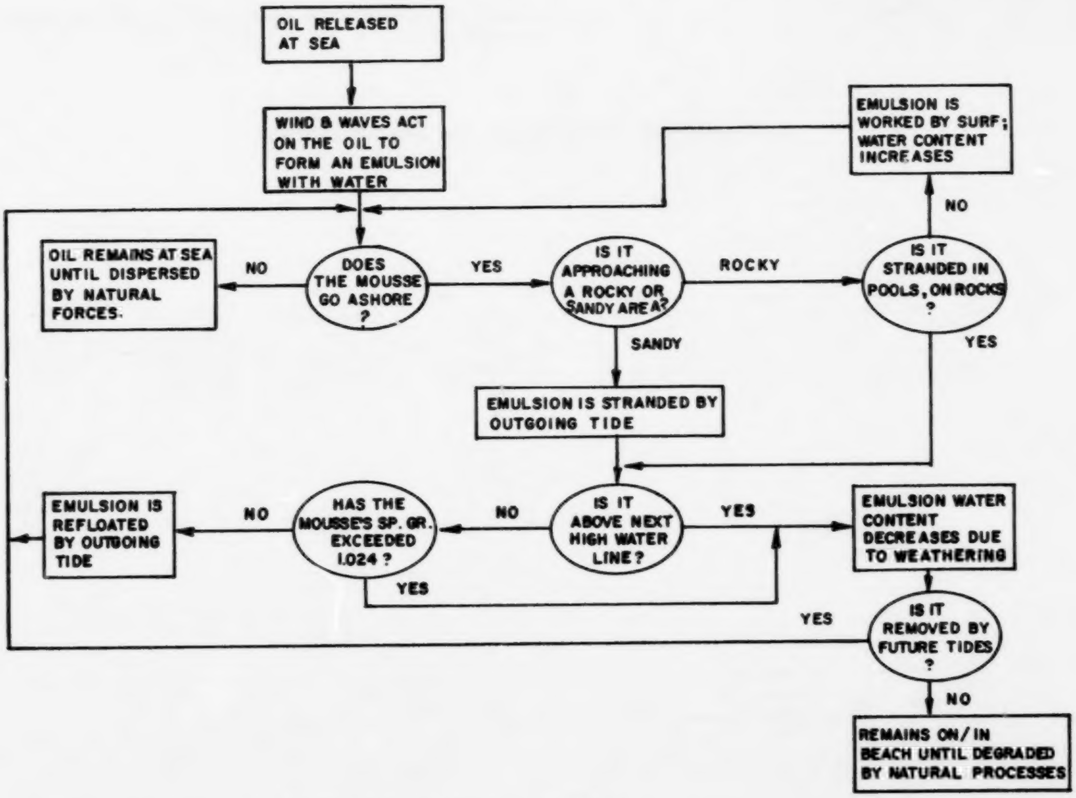
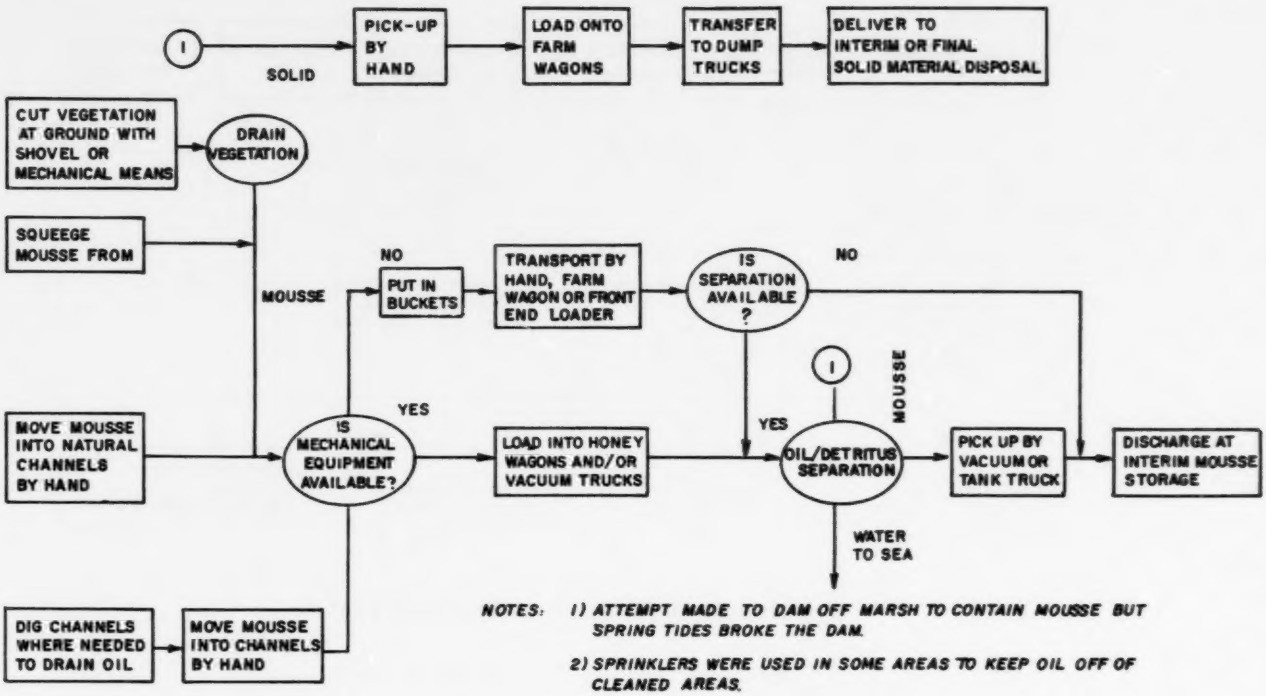
APPENDIX


SECTEUR de
SECTEUR de ST. RENAN
SECTEUR de LANNILIS
SECTEUR de ROSCOFF
SECTEUR de PLOUGASNOU
SECTEUR de LANNION
SECTEUR de ST. BRIEUC
AT SEA ACTIVITIES
FINISTERE COAST
FINISTÈRE COAST
P.C. POL MAR MARINE
P.C. PLOUDALMEZEAU FINISTERE
P.C. ZONE RENNES COTES-DU-NORD
STUDIES CHEMISTRY EVALUATION OF METHODS
PORT ACTIVITIES
LOGISTICS & METHODS
CONSTRUCTION OF ULTIMATE SOLID WASTE STORAGE SYSTEMS

Approximate organization of the response effort.


OIL RELEASED AT SEA
WIND & WAVES ACT ON THE OIL TO FORM AN EMULSION WITH WATER
EMULSION IS WORKED BY SURF; WATER CONTENT INCREASES
NO
OIL REMAINS AT SEA UNTIL DISPERSED BY NATURAL FORCES.
NO
DOES THE MOUSSE GO ASHORE ?
YES
IS IT APPROACHING A ROCKY OR SANDY AREA?
ROCKY
IS IT STRANDED IN POOLS, ON ROCKS ?
SANDY
YES
EMULSION IS STRANDED BY OUTGOING TIDE
EMULSION IS REFLOATED BY OUTGOING TIDE
NO
HAS THE MOUSSE'S SP. GR. EXCEEDED 1.024 ?
NO
IS IT ABOVE NEXT HIGH WATER LINE?
YES
EMULSION WATER CONTENT DECREASES DUE TO WEATHERING
YES
YES
IS IT REMOVED BY FUTURE TIDES ?
NO
REMAINS ON/IN BEACH UNTIL DEGRADED BY NATURAL PROCESSES

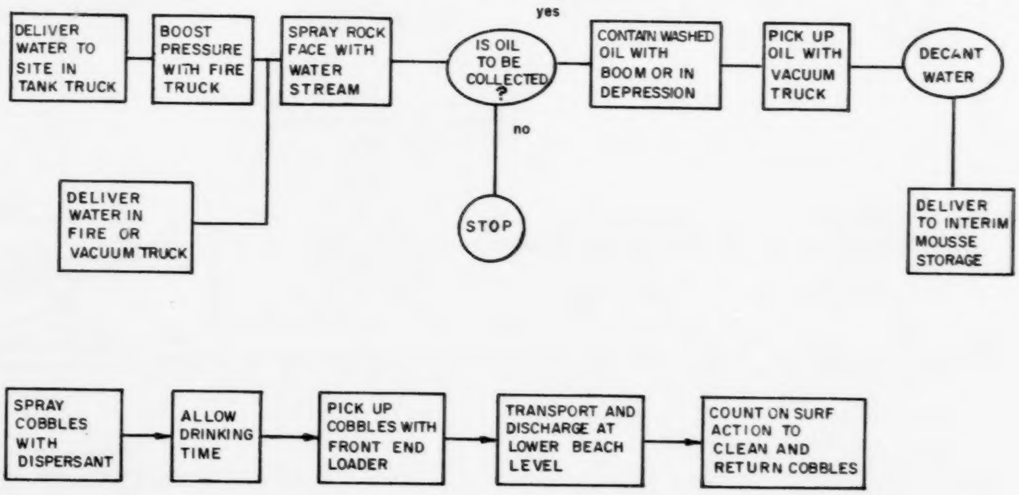
Possible pathways of oil spilled from Amoco Cadiz.


1
SOLID
PICK-UP BY HAND
LOAD ONTO FARM WAGONS
TRANSFER TO DUMP TRUCKS
DELIVER TO INTERIM OR FINAL SOLID MATERIAL DISPOSAL
CUT VEGETATION AT GROUND WITH SHOVEL OR MECHANICAL MEANS
DRAIN VEGETATION
SQUEEGE MOUSSE FROM
MOUSSE
NO
PUT IN BUCKETS
TRANSPORT BY HAND, FARM WAGON OR FRONT END LOADER
IS SEPARATION AVAILABLE ?
NO
1
MOUSSE
YES
MOVE MOUSSE INTO NATURAL CHANNELS BY HAND
IS MECHANICAL EQUIPMENT AVAILABLE?
LOAD INTO HONEY WAGONS AND/OR VACUUM TRUCKS
YES
OIL/DETRITUS SEPARATION
PICK UP BY VACUUM OR TANK TRUCK
DISCHARGE AT INTERIM MOUSSE STORAGE
WATER TO SEA
DIG CHANNELS WHERE NEEDED TO DRAIN OIL
MOVE MOUSSE INTO CHANNELS BY HAND

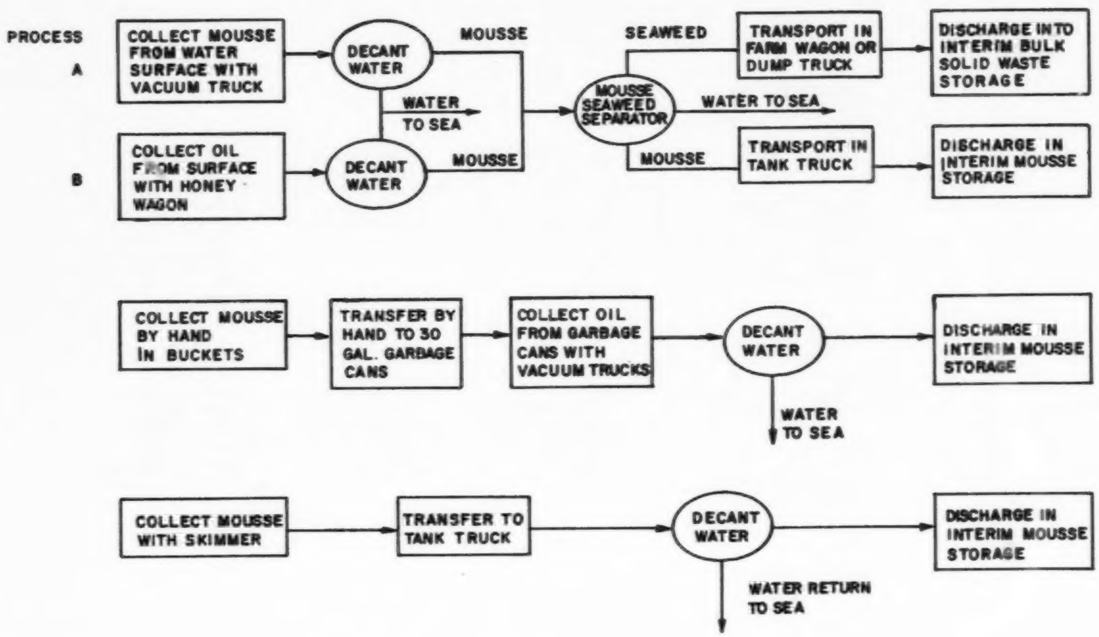
NOTES: 1) ATTEMPT MADE TO DAM OFF MARSH TO CONTAIN MOUSSE BUT SPRING TIDES BROKE THE DAM.

2) SPRINKLERS WERE USED IN SOME AREAS TO KEEP OIL OFF OF CLEANED AREAS.

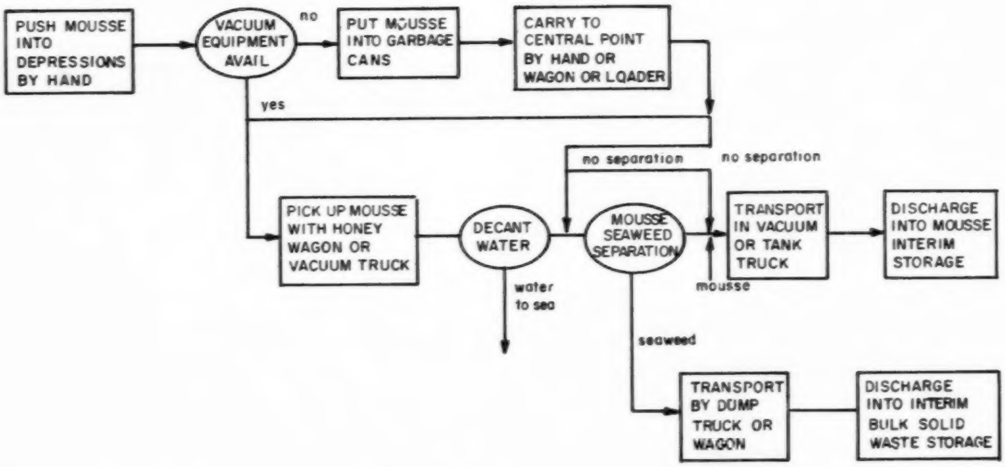
Unit process: Cleaning of Ile Grande marsh.


DELIVER WATER TO SITE IN TANK TRUCK
BOOST PRESSURE WITH FIRE TRUCK
SPRAY ROCK FACE WITH WATER STREAM
yes
IS OIL TO BE COLLECTED ?
CONTAIN WASHED OIL WITH BOOM OR IN DEPRESSION
PICK UP OIL WITH VACUUM TRUCK
DECANT WATER
no
DELIVER WATER IN FIRE OR VACUUM TRUCK
STOP
DELIVER TO INTERIM MOUSSE STORAGE
SPRAY COBBLES WITH DISPERSANT
ALLOW DRINKING TIME
PICK UP COBBLES WITH FRONT END LOADER
TRANSPORT AND DISCHARGE AT LOWER BEACH LEVEL
COUNT ON SURF ACTION TO CLEAN AND RETURN COBBLES

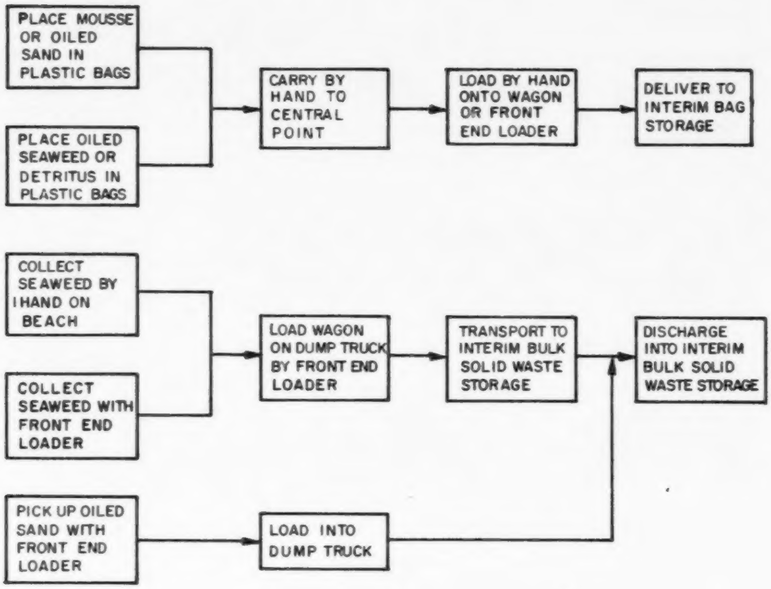
Unit process: Cleaning of walls, rockfaces, and cobbles.


PROCESS
A
COLLECT MOUSSE FROM WATER SURFACE WITH VACUUM TRUCK
DECANT WATER
MOUSSE
WATER TO SEA
B
COLLECT OIL FROM SURFACE WITH HONEY WAGON
DECANT WATER
MOUSSE
MOUSSE SEAWEED SEPARATOR
SEAWEED
TRANSPORT IN FARM WAGON OR DUMP TRUCK
DISCHARGE INTO INTERIM BULK SOLID WASTE STORAGE
WATER TO SEA
MOUSSE
TRANSPORT IN TANK TRUCK
DISCHARGE IN INTERIM MOUSSE STORAGE
COLLECT MOUSSE BY HAND IN BUCKETS
TRANSFER BY HAND TO 30 GAL. GARBAGE CANS
COLLECT OIL FROM GARBAGE CANS WITH VACUUM TRUCKS
DECANT WATER
WATER TO SEA
DISCHARGE IN INTERIM MOUSSE STORAGE
COLLECT MOUSSE WITH SKIMMER
TRANSFER TO TANK TRUCK
DECANT WATER
WATER RETURN TO SEA
DISCHARGE IN INTERIM MOUSSE STORAGE

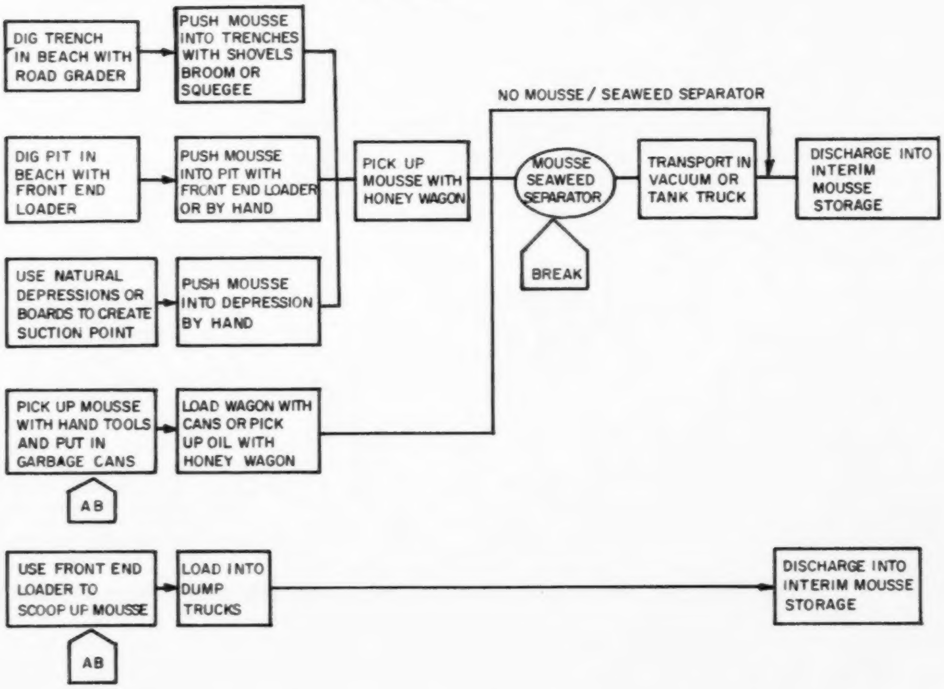
Unit process: Removal of mousse from water surface to interim storage.


PUSH MOUSSE INTO DEPRESSIONS BY HAND
VACUUM EQUIPMENT AVAIL
no
PUT MOUSSE INTO GARBAGE CANS
CARRY TO CENTRAL POINT BY HAND OR WAGON OR LOADER
yes
no separation
no separation
PICK UP MOUSSE WITH HONEY WAGON OR VACUUM TRUCK
DECANT WATER
water to sea
MOUSSE SEAWEED SEPARATION
mousse
seaweed
TRANSPORT IN VACUUM OR TANK TRUCK
DISCHARGE INTO MOUSSE INTERIM STORAGE
TRANSPORT BY DUMP TRUCK OR WAGON
DISCHARGE INTO INTERIM BULK SOLID WASTE STORAGE

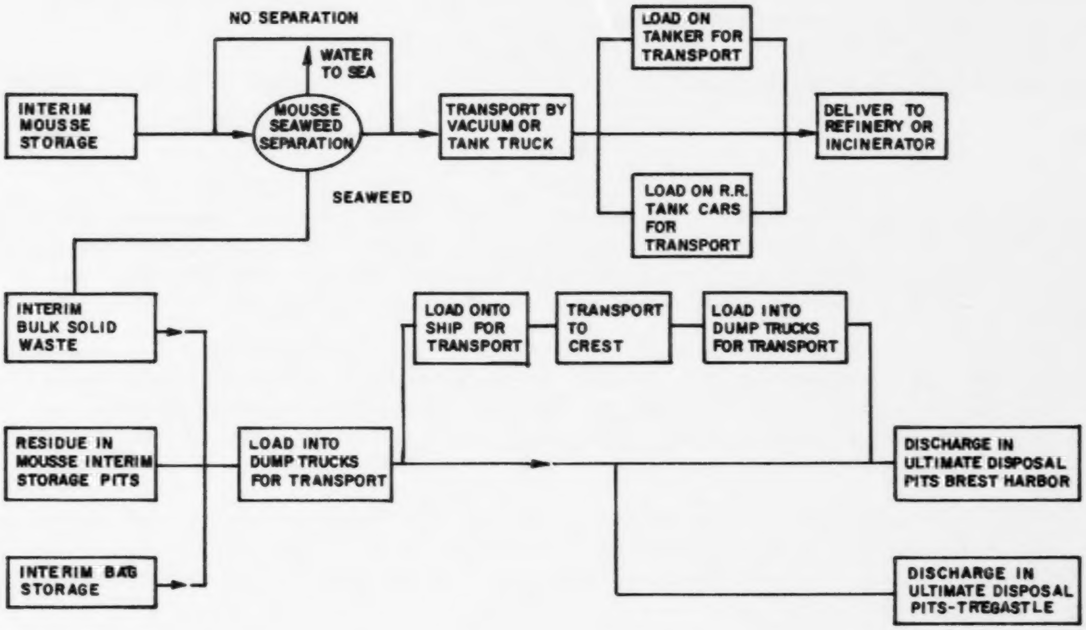
Unit process: Removal of mousse from rocky areas to interim storage.


PLACE MOUSSE OR OILED SAND IN PLASTIC BAGS
PLACE OILED SEAWEED OR DETRITUS IN PLASTIC BAGS
CARRY BY HAND TO CENTRAL POINT
LOAD BY HAND ONTO WAGON OR FRONT END LOADER
DELIVER TO INTERIM BAG STORAGE
COLLECT SEAWEED BY HAND ON BEACH
COLLECT SEAWEED WITH FRONT END LOADER
LOAD WAGON ON DUMP TRUCK BY FRONT END LOADER
TRANSPORT TO INTERIM BULK SOLID WASTE STORAGE
DISCHARGE INTO INTERIM BULK SOLID WASTE STORAGE
PICK UP OILED SAND WITH FRONT END LOADER
LOAD INTO DUMP TRUCK

Unit process: Removal of oiled sand, seaweed, and detritus to interim storage.


DIG TRENCH IN BEACH WITH ROAD GRADER
PUSH MOUSSE INTO TRENCHES WITH SHOVELS BROOM OR SQUEGEE
NO MOUSSE / SEAWEED SEPARATOR
DIG PIT IN BEACH WITH FRONT END LOADER
PUSH MOUSSE INTO PIT WITH FRONT END LOADER OR BY HAND
PICK UP MOUSSE WITH HONEY WAGON
MOUSSE SEAWEED SEPARATOR
BREAK
TRANSPORT IN VACUUM OR TANK TRUCK
DISCHARGE INTO INTERIM MOUSSE STORAGE
USE NATURAL DEPRESSIONS OR BOARDS TO CREATE SUCTION POINT
PUSH MOUSSE INTO DEPRESSION BY HAND
PICK UP MOUSSE WITH HAND TOOLS AND PUT IN GARBAGE CANS
AB
LOAD WAGON WITH CANS OR PICK UP OIL WITH HONEY WAGON
USE FRONT END LOADER TO SCOOP UP MOUSSE
AB
LOAD INTO DUMP TRUCKS
DISCHARGE INTO INTERIM MOUSSE STORAGE

Unit process: Removal of stranded mousse from beach to interim storage.


NO SEPARATION
WATER TO SEA
INTERIM MOUSSE STORAGE
MOUSSE SEAWEED SEPARATION
TRANSPORT BY VACUUM OR TANK TRUCK
LOAD ON TANKER FOR TRANSPORT
DELIVER TO REFINERY OR INCINERATOR
SEAWEED
LOAD ON R.R. TANK CARS FOR TRANSPORT
INTERIM BULK SOLID WASTE
LOAD ONTO SHIP FOR TRANSPORT
TRANSPORT TO CREST
LOAD INTO DUMP TRUCKS FOR TRANSPORT
RESIDUE IN MOUSSE INTERIM STORAGE PITS
LOAD INTO DUMP TRUCKS FOR TRANSPORT
DISCHARGE IN ULTIMATE DISPOSAL PITS BREST HARBOR
INTERIM BAG STORAGE
DISCHARGE IN ULTIMATE DISPOSAL PITS-TREGASTLE

Unit process: Movement of mousse and oiled material from interim storage to disposal.